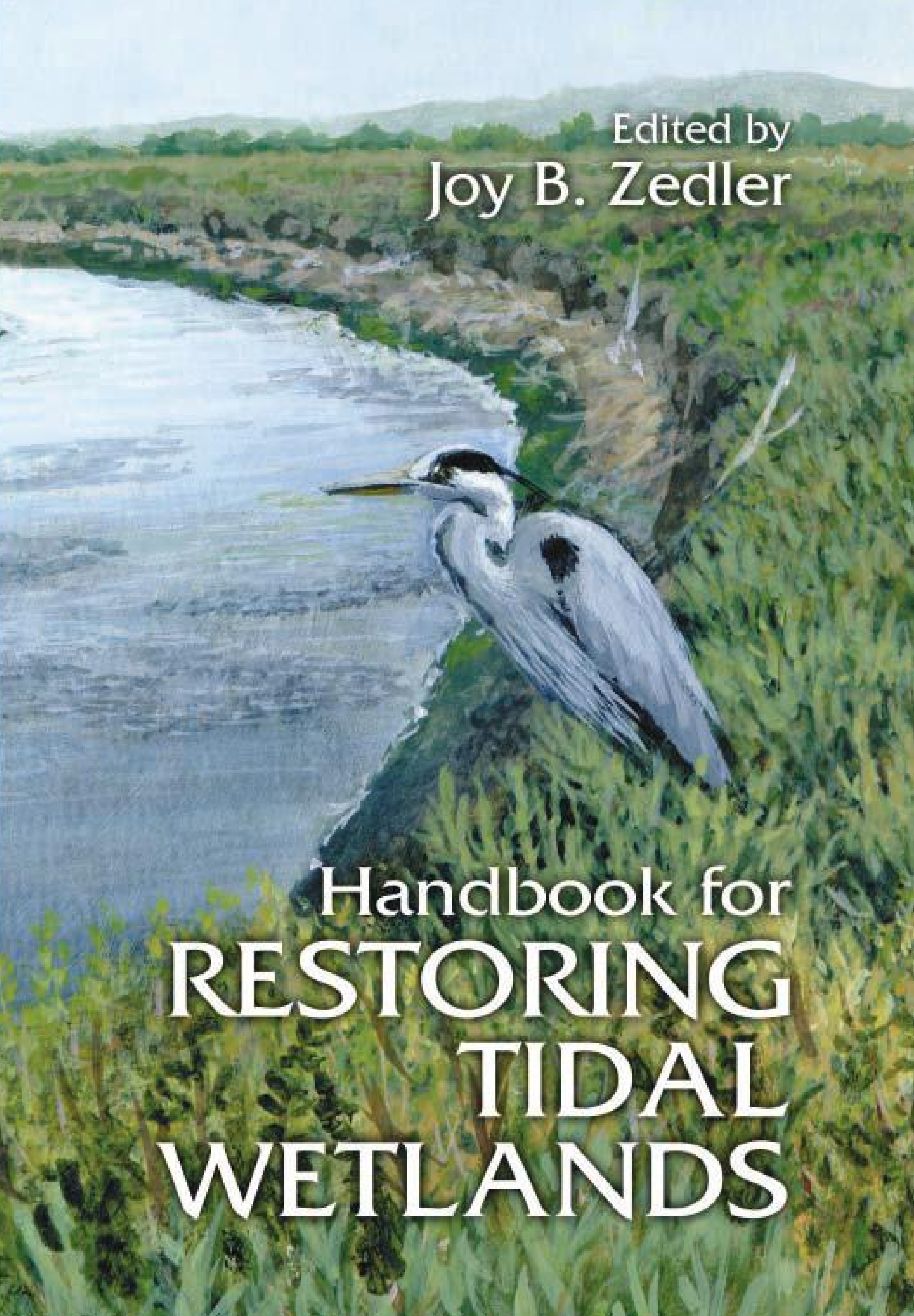

Edited by
Joy B. Zedler

Handbook for RESTORING TIDAL WETLANDS

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Dedication

We dedicate this book to all those who have helped protect coastal wetlands in southern California and in northern Mexico — the people who had the courage and stamina to say, over and over, "Enough development! We need these last wetland remnants for open space, for wildlife, and for our grandchildren."

The list of persons who have conserved coastal wetlands includes numerous personnel from resource agencies, such as the U.S. Fish and Wildlife Service, California Coastal Commission, and California State Coastal Conservancy. It also includes several members of the public who deserve special mention:

For Tijuana Estuary, we thank Mike and Pat McCoy, who convinced the U.S. Fish and Wildlife Service to purchase the best part of the estuary, thereby saving it from becoming a marina. While decision-makers were calling the estuary a "swill pit" because it suffered from raw sewage spills, Mike and Pat campaigned endlessly for its protection as an endangered species refuge. They were right. The sewage is now treated and diverted, and the rare species persist.

For Famosa Slough, we thank Jim and Barbara Peugh, who formed a citizen action group and convinced the City of San Diego to purchase this 8-hectare privately owned wetland that was cut off from Mission Bay by the San Diego River Flood Control Channel. Jim and Barbara countered the many tricks of the developer, who first declared that the wetland was illegal because unknown persons were blocking open the tide gates, who posted a guard to keep the tide gates closed, expecting the wetland to dry up, and who then proposed to fill half the site and call the housing development "The Sanctuary." Thanks to the Friends of Famosa Slough, the entire wetland remnant is now a *real* sanctuary.

For Sweetwater Marsh in San Diego Bay, we thank Joan Jackson, who led the effort to improve mitigation for highway widening and flood control channel construction by increasing the area of land set aside for the public, by ensuring that 12-story hotels were not built on Gunpowder Point, by preventing further fragmentation of the remnant salt marsh by new roads, and by raising the standards for habitat loss compensation.

For Los Peñasquitos Lagoon, we also thank Joan Jackson, whose vigilance as head of the LPL Foundation has minimized encroachments by a sewer pump station and pipelines, a highway interchange, street widening, and urban runoff. Urban wetland remnants are continually besieged with proposals to permit just one more project that will destroy "less than 5% of the wetland," without acknowledging the cumulative impacts of a dozen such projects.

For establishing the nonprofit organization, pro esteros, and for heroic efforts to preserve wetlands in Mexico, we thank Barbara Massey and Silvia Ibarra-Obando. For the consortium of conservation groups known as "Save California Wetlands" and for tireless efforts to improve the scientific basis of wetland restoration plans, we thank Marcia Hanscom.

We dedicate this book to all the conservation leaders who have worked to hold the line in a region with rapid population growth and increasingly vulnerable wetland remnants.

Hundreds of concerned citizens have joined these leaders in the effort to manage more wisely the inevitable development of southern California. Their efforts serve as models for the thousands who will need to steward these lands in the future and who will need to add other threatened habitats if this region's high biological diversity is to be sustained. Their progress shows that *individuals can make a difference*!

Joy B. Zedler
Madison, Wisconsin

Preface

Ecosystem restoration has been practiced for many decades, and much has been written about the process; yet few texts or handbooks are available to guide the planning, implementation, or assessment phases. The practitioner must consult a variety of sources to learn what environmental conditions to pay attention to and which species to consider reintroducing. Anecdotes and photographs may be all that are available to indicate the techniques that work best and the pitfalls that can be avoided. Project managers and resource agency staffs face a similar dilemma: scientific papers and project reports treat single sites or specific taxa, but there is no broad-based compilation of case studies and principles to guide the management of restoration sites.

As affiliates of the Pacific Estuarine Research Lab (PERL), based at San Diego State University, the authors of this book have all accumulated expertise that bears on these needs. We wrote this handbook so that our knowledge would fill some of the information gaps in restoring coastal wetlands, particularly the tidal ecosystems of southern California. Our intended audience includes *planners* of projects to restore, enhance, and construct coastal wetlands; *practitioners* who implement those plans; *resource managers* who oversee the sites; and *students* of restoration ecology. In addition, we hope that *researchers* will use the literature reviews to identify knowledge gaps and that concerned *members of the public* will find information of value to future conservation efforts.

We hope the general concerns and advice expressed in this book will be useful to all organizations and individuals who are involved in coastal wetland restoration. We have drawn from the literature on coastal restoration across the U.S., while restricting our presentation of details to examples from coastal wetlands along the Pacific coast of southern California and northern Baja California. The rapid growth of the human population in this region has accelerated impacts to coastal wetlands. As a result, the policy of requiring compensation for unavoidable wetland losses has generated many large habitat construction projects, which have had variable and often unpredictable outcomes. Urban expansion and compensatory mitigation are not unique to our region, nor is the need to be proactive in considering how to restore and replace damaged ecosystems. The time is right for a handbook to guide future mitigation and restoration projects around the nation. The opportunity to write one came from our long-term study of restoration efforts in this region and the encouragement of the series editor, Michael Kennish, and the CRC Press LLC editor, John Sulzycki.

Our text is enhanced by Donovan McIntire's original illustrations, developed especially to aid plant and animal species identification in these coastal ecosystems. The illustrations include organisms that are poorly known but have utility in the assessment process (plants, fishes, and invertebrates). Birds are well illustrated in many field guides; they are not emphasized here.

We are grateful to many people who have facilitated and encouraged our research over the years. First, we thank colleagues who served on thesis committees and gave helpful advice, and we thank the many PERL students, staff, and volunteers who, over

the past 15 years, trekked through the mud and water to collect basic ecological data. Second, we are grateful to several agencies for access to study sites. It has been our good fortune to study several restoration efforts, with funding from the U.S. Fish and Wildlife Service, the California Sea Grant College Program, the California Department of Transportation, the Coastal Ocean Program of the National Oceanic and Atmospheric Administration (NOAA), the NOAA Sanctuaries and Reserves Division's National Estuarine Research Reserve Program, the U.S. Geological Survey Biological Resources Division, Los Peñasquitos Lagoon Foundation, the National Science Foundation, and the Earth Island Institute. The latter organization not only made a long-term research program possible, it also supported the writing effort. Finally, we thank our reviewers who were generous with their time and ideas; their comments improved each of the chapters and helped broaden the scope of the book:

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While computers eliminated the need for a separate typist, they could not replace the keystone of our writing team. We thank Janelle West for keeping track of the pieces, sending drafts around the world, fixing the problems, and managing the paperwork.

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Zedler's research interests include restoration and wetland ecology, role of biodiversity in ecosystem function, use of mesocosms in wetland research, invasive plants, and adaptive management. She helps edit four peer-reviewed journals (*Ecological Applications*, *Wetlands Ecology and Management*, *Ecological Engineering*, *Ecosystems*) and is a recent recipient of the national Wetlands Award for Science Research (Environmental Law Institute and Environmental Protection Agency). She is a member of The Nature Conservancy Governing Board; the Environmental Defense Board of Trustees; Wisconsin Natural Areas Preservation Council, and several professional societies (Ecological Society of America, Estuarine Research Federation, Society for Ecological Restoration, and Society of Wetland Scientists). As Director of Research for the Arboretum, she is developing a Center for Restoration Ecology that seeks to improve restoration effectiveness.

chapter one

Introduction

Joy B. Zedler

1.1 The scope of this book

This is a book about restoring tidal marshes, with broad geographic treatment and detailed examples from southern California. To begin, we ask a very general question: "What constrains ecosystem restoration efforts?" In answering it, we look to the literature and the theory of restoration ecology; we then explore what our study sites (Boxes 1.1 to 1.12) bring to bear on theory, as well as where they fit in a broad spectrum of restoration projects. Next, we consider the special concerns of coastal wetland restoration and salt marshes in particular. My conclusion in Chapter 1 is that adaptive approaches (and more examples of how to incorporate science into restoration) are sorely needed.

In Chapter 2, Gabrielle Vivian-Smith brings her experience from Australia, New Jersey, and southern California to bear on how to set restoration goals and identify reference ecosystems. Because so many restoration projects have insufficient spatial and topographic heterogeneity, she emphasizes the importance of including and connecting different habitat types, incorporating tidal creek networks into salt marshes, and adding microtopographic variability.

In Chapter 3, John Callaway characterizes wetland topography and soils, based on his broad knowledge of the literature, as well as his experience in San Francisco Bay, Louisiana, Europe, and southern California. His focus is on the dynamic processes that affect restoration sites. While there is still much to learn about manipulating hydrology and soils in restoration sites, experiences in southern California and elsewhere make it clear that the restored wetlands will not match the functioning of natural wetlands unless hydrology and soils are properly restored. Too little tidal action and too much river influence will create a brackish ecosystem. Soil that is too coarse, too low in nutrients, and depauperate in organic matter will prevent optimal plant growth and reduce habitat potential for animals. Because erosion and sediment accretion are not yet predictable, restoration plans need to include contingencies and, again, an adaptive approach.

Gary Sullivan's contribution on vegetation (Chapter 4) draws on a wealth of information from the literature, as well as his experiences in New York, southern California, and Oregon wetlands. Methods of salvaging, holding, propagating, and transplanting salt marsh plants are described in detail, supported by new data from field and greenhouse experiments.

Fishes and invertebrates are the focus of Chapter 5 prepared by Greg Williams and Julie Desmond. Both have worked extensively in coastal wetlands — Greg in Washington

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and both in southern California. Their long-term efforts to collect and understand monitoring data from three southern California coastal wetlands provide a solid background for their many recommendations on how to improve salt marsh restoration.

Chapter 6 is a compilation of methods we use in initial site surveys, monitoring, and ecosystem assessment, updated from our earlier assessment manual (PERL 1990). New information is presented on sedimentation methods, global positioning systems, geographic information systems, and remote sensing, added by John Callaway, Gary Sullivan, Julie Desmond and Greg Williams.

In Chapter 7, John Callaway and Gary Sullivan describe their collective experiences with restoration problems and the need to plan for maintenance and long-term management. Again, we emphasize the need for adaptive approaches.

Additional contributions are those of Janelle West, who described the various sites we treat throughout the book; Sharook Madon, an expert in fish bioenergetics, who has worked on both the Atlantic and Pacific Coasts; Gregory Noe, whose research on the annuals of the high marsh calls attention to the problems of exotic species and complex germination cues; Meghan Fellows, who provided recommendations from her research on the reintroduction of an endangered salt marsh plant; and Matt James, who added recommendations from his recent research in the wetland-upland ecotone.

Donovan McIntire prepared more than 80 new illustrations of plants and animals that inhabit our region's coastal wetlands. Their accuracy and completeness increase the utility of this book, while their beauty enhances the publication. McIntire is an expert, observant naturalist with detailed knowledge of both the plants and animals of the salt marsh. Wherever we mention the plants, fishes, and invertebrates, we encourage readers to consult these illustrations.

In each chapter, we take a national perspective, but we elaborate only on the ecosystems we have studied and observed, some for over 20 years. To gain predictability, we need detailed documentation of restoration sites of many kinds in many places. We need case studies to describe conditions across the broad restoration spectrum (Section 1.4.2). Ultimately, with appropriate long-term documentation for a variety of ecosystems of similar type, it should be possible to test more general theories of how restored ecosystems develop over time and to make ecological restoration more scientific — that is, more predictable.

1.2 The shortcomings of restoration ecology theory

Efforts to direct the recovery of damaged sites and landscapes date back at least as far as Aldo Leopold's work in the Midwest in the 1930s (Leopold 1949), but the term "restoration ecology" dates to a conference held in 1984 (Jordan et al. 1985). More formal establishment of restoration as a branch of ecology came about with the development of the journal *Restoration Ecology* (1993 ff.). Because restoration ecology is young, and because restoration is scientifically challenging, we still have much to learn. Ecosystems are extremely difficult to restore in full, because they have so many components and support very complex interactions. We need better understanding of cause-effect relationships, both between the abiotic and biotic components and among species.

If scientists fully understood the conditions and controlling variables at restoration sites, they would be able to predict the outcomes of restoration efforts. Promises could then be made and kept. If there were no constraints, practitioners could plant the restoration site and walk away. Sites could be planted and follow-up would not be needed. Of course, neither is the case. Understanding is incomplete, and restoration efforts are constrained by insufficient knowledge and by site-to-site variability. If we had intimate knowledge of how ecosystems develop in every restoration setting, we could simulate outcomes

of alternative restoration efforts. In reality, no one can guarantee if or when restoration goals will be met or exactly what actions it will take to make a site comply with expectations by a specific date.

1.2.1 The status of restoration theory

Restoration ecologists are beginning to use restoration sites to test various theories derived from both natural and human-disturbed ecosystems. For the most part, however, practitioners have had to rely on ideas developed in the broader field of ecology and tested in very different settings. Although several ecological theories are relevant to restoration (Section 1.2.2) and several models have been proposed to describe ecosystem development in restoration sites (Section 1.2.3), one concept — that of succession — dominates theoretical considerations (Section 1.2.4). Individual sites are expected to follow some orderly pathway of development over time. In the absence of sufficient long-term data on how restoration sites actually develop, there is little recognition of the complexity of processes that determine pathways and outcomes.

In a recent paper, Joan Ehrenfeld (2000) suggests that restoration ecologists should stop expecting to find simple rules or Newtonian laws that can predict restoration outcomes. Instead, we should acknowledge the diversity of challenges posed by both the sites themselves and the goals we set for them. Discussion of her ideas at The Environmental Defense Fund's May 1999 Science Day centered around the need for demonstration sites where alternative restoration efforts would be tested in an experimental framework — an approach that I call "adaptive restoration" (Section 1.5).

1.2.2 Relevant ecological theories

A special issue of *Restoration Ecology* discusses the need for a conceptual basis for this area of science and offers suggestions from population, community, and ecosystem perspectives. The ideas of several authors (Ehrenfeld and Toth 1997, Montalvo et al. 1997, Palmer et al. 1997) are discussed below, in relation to those of others who have called for a stronger theoretical base in restoration ecology. The list of relevant theories goes well beyond succession.

At the population level, Montalvo et al. (1997) call for research on genetic variation, life history traits, metapopulation dynamics, effects of selection, and interspecific interactions in restoration settings. The hypothesis that genetic diversity is essential to population persistence has special relevance because several plants that are used widely in restoration projects (species of *Zostera* and *Spartina*) are propagated from one or a few clones. In another journal, and with a more practical perspective, van Andel (1998) reviews transplant studies of genetic vs. environmental effects on plant growth and fecundity and considers how genetic theories might be applied to restoration. He asks whether propagules should come from single or multiple sources and argues that it is always safer to include mixed material, as selection can then act on the larger gene pool. The risks of introducing material with too little variation and of introducing unwanted genotypes, he says, are minimal. In general, he finds insufficient evidence for strong conclusions, so long as transplants are moved about within climatic and soil types, and much leeway is given for "trial and error approach(es)" in restoration projects.

At the community and ecosystem level, Palmer et al. (1997) identify several hypotheses that are of direct interest to restoration: (1) Ecosystems are not at equilibrium (this poses problems for setting specific endpoints); (2) Diversity affects community and ecosystem stability (if so, then more species should be introduced); (3) Individual species matter to ecosystem function (if so, then we need to know which species play key roles); (4) Habitat

heterogeneity produces biodiversity (the authors note that this relationship is largely untested in restoration contexts); and (5) Natural disturbance regimes must be restored. Kenneth Potter (University of Wisconsin, *personal communication*) argues an alternative, that restoring some portion of the natural range of disturbance can support ecosystem structure and function (this debate is pertinent to the question of how much freshwater flooding salt marshes require). Ehrenfeld and Toth (1997) emphasize the need to consider ecosystem functions, and Bell et al. (1997) recommend that restorationists take a landscape approach, without specifying the theories that should guide restoration efforts.

In other publications, MacMahon (1998), Cole (1999), and Middleton (1999) also try to identify the role of theory in restoration. MacMahon (1998) explores the importance of facilitation as a restoration tool. His work in desert ecosystems supports the hypothesis that plants placed in clusters facilitate one another's survival and growth. Hence, restoration should proceed more rapidly if compatible groupings can be identified and established than if the same plants are spread evenly across the site. Sites with stressful conditions probably benefit most from cluster plantings, so the "nurse plant" concept has merit for salt marshes. Indeed, in New England, Bertness and Shumway (1993) show that shade plants reduce soil salinities and facilitate the establishment of halophyte seedlings.

Cole (1999) begins by listing theories that have some bearing on wetland restoration. I have reworded his collection as follows: (1) Habitats (physical space) consist of niches (abstract space that defines species' roles); (2) Disturbance regimes alter plant communities; (3) Large- and small-scale disturbances affect ecosystems differently; (4) Competition among species influences the assemblages that persist; (5) Islands of habitat that are small and isolated will support fewer species; and (6) Successional pathways are influenced in part by starting conditions and specific processes (facilitation, tolerance, and inhibition). Middleton's (1999) list is similar, but the emphasis of her book is on the importance of flood pulsing to river and riparian restoration projects.

It would appear that many theories are relevant to restoration ecology. Indeed, any concept that helps explain species distribution or abundance has relevance. Because restoration projects are so often expected to produce fast results, to justify funding or meet mitigation requirements, ecosystem development over time is of greatest interest. Perhaps that is why conceptual models focus on the temporal aspects of restoration.

1.2.3 Models of restoration site development

Ecological theory in urban and disturbed settings lags far behind the need for predictability. As ecologists try harder to become predictive, their models become more complex. But few researchers have obtained long-term data from restoration sites. Hence, a simple model of the restoration process may be all that is possible — it certainly is all that is available. The models of Kentula et al. (1992), Hobbs and Mooney (1993), and Richardson (1994) plot hypothesized performance curves (ecosystem attribute × time). The models differ in the details — whether or not the target might be missed, whether or not variability in pathways or outcomes is illustrated (e.g., by error bars), and whether or not alternative pathways might be expected (different shaped curves).

Simple ecological models are inadequate almost by definition, and models of restoration performance are no exception (see review in Zedler and Callaway 1999). But many practitioners and decision-makers still cling to the idea that we can predict restoration outcomes rather precisely. Every time a regulatory agency agrees to a mitigation project, a simple model is endorsed: if project x is conducted at site y, it will replace the functions and values lost at site z.

One model by Bradshaw (Bradshaw 1984, 1985, 1997, Dobson et al. 1997) is the most often cited (e.g., see Meffe et al. 1997 and MacMahon 1998). Rather than plotting performance over time, this model predicts a linear relationship between structure (x axis) and function (y axis) as a degraded site develops along a straight-line trajectory toward its target, the original, predisturbance ecosystem. A linear relationship is not supported by experimental data, although few data are available to compare structure and function in any ecological setting (Zedler and Lindig-Cisneros *in press*). A recent surge of experiments has sought to test the relationship of ecosystem function (e.g., productivity, invasibility) and a common attribute of structure, namely plant species diversity (Tilman and Downing 1994, Naeem et al. 1996, Hooper and Vitousek 1997, Tilman 1997, Tilman et al. 1997). It is now clear that ecosystem functioning is affected not only by the diversity of plant species, but also the presence of various plant functional groups (guilds) and specific types of species (Tilman et al. 1997).

Because restoration ecology is a young science, we do not yet have a thick catalog of case studies or restoration experiments to use in testing these models — not for many individual sites and certainly not for many types of sites. Our survey of 26 peer-reviewed papers that concern tidal marsh restoration (Zedler and Callaway *in press*) included a broad range of situations, but most studies were based on small sites, studied for short periods of time, and involved only a few ecosystem attributes. No papers were found describing the long-term evaluation of large sites with measures of change in multiple attributes (topography, hydrology, soils, vegetation, and fauna). The common assumption that sites should develop along pathways familiar to secondary succession needs to be re-evaluated.

1.2.4 Do restoration projects follow succession theory?

If a site is disturbed and allowed to recover without intervention, we consider the process to be "secondary succession." An early paper on secondary succession (Keever 1950) explained the sequence of invasions by old-field species in North Carolina on the basis of life-history traits (early colonists being those with many seeds that had no dormancy requirement), allelopathy (one species that changed the environment so that its own seeds had reduced viability), and shading (shrubs and saplings outcompeting understory species). Ecologists spent the next 50 years developing additional case studies and experiments to test these and other ideas about the pathways of individual sites. The processes that are now known to affect the outcome include competition and facilitation among community members.

The general expectations for terrestrial ecosystems are that herbaceous vegetation will appear first and that shrub and tree layers will appear later, if the climate can support them. But there is still limited predictability about which species will appear and when and how they will expand and decline. A typical ecology textbook will present one or two examples, which are acknowledged as being site specific. It is widely understood that secondary succession is complicated — what develops on a site will be a function of which organisms can disperse to the site, which species establish first, which animals affect the vegetation, what environmental conditions happen to be present at the time of establishment and thereafter, and what disturbance regimes follow. Predictability is of the most general sort — if a pine forest blows down in 1999, we might expect a pine forest to reestablish by 2030. But few would venture to guarantee the number of trees of each species, let alone the population sizes of individual understory plants or animals.

Contrast this acknowledgment of complexity with attitudes about the ecosystem restoration process. In the context of compensatory mitigation, a developer plans to fill a

wetland and is required to replace it through restoration or creation of a wetland somewhere else. A promise is made that what is lost will be replaced, or that a larger area of an alternative type of wetland will be provided to compensate for the damages, if in-kind compensation is too difficult. The developer or his/her consultant predicts that what is present in one place can be replaced 1:1 (hectare for hectare) at another place; if not, then an area of 2:1 or 3:1 of something slightly different will replace the area lost. Such predictions are usually made before a mitigation site has even been chosen. Hence, the "promise" makes a huge leap beyond our already inadequate theory of secondary succession.

There are two reasons why existing succession theory cannot be so easily extended. First, we rarely know much about the mitigation site — so how can we assume it will be able to provide the desired structural and functional attributes? Second, the habitat restoration or creation project will not be left to develop on its own, as in secondary succession; rather, practitioners will attempt to accelerate the process by engineering the topography and hydrology, amending the soil, and/or manipulating the vegetation and, in a few cases, the fauna. So, why should the site be expected to follow some desired pathway? In restoration contexts, ecologists are expected to predict outcomes under conditions that are far more complex than for secondary succession and under conditions that are unlike those of an abandoned field. Restorationists draw on "succession theory," but without adequate tests of how well it fits restoration scenarios. Furthermore, the sites are often highly degraded. Much of the funding for restoration in the mitigation context takes place in urban areas, which are poorly understood ecologically. It is noteworthy that the National Science Foundation has only recently funded Long-Term Ecological Research in urban settings; unfortunately, there were only enough funds to support such research for Baltimore and Phoenix. The findings of these two projects may well provide insights of great importance to restoration ecology, if only because long-term data will be gathered in highly disturbed settings.

1.3 The lack of predictability of restoration outcomes

Experiences in southern California suggest that the pathway taken by a restoration site is not highly predictable. In fact, our graphical plots of several variables measured for ten years at San Diego Bay gave conflicting results — soil organic matter leveled off, total Kjeldahl nitrogen showed a slow but linear increase, and the height of cordgrass vegetation declined over time (Zedler and Callaway 1999). Thus, one could predict that the system had stabilized, or that it was improving, or that it was diverging away from the target, depending on the variable inspected. Furthermore, we encountered many surprises in our studies of restoration projects. Obviously, a full understanding of the cause-effect relationships among ecosystem components is needed, and that came about only after long-term study and experimentation at multiple scales.

Carpenter et al. (1995) emphasize the importance of evaluating the responses of whole ecosystems, using not just micro- or mesocosm scale experiments, but whole-system experiments. They offer many examples of different outcomes in experimental treatments at small and large scales. Similarly, in San Diego Bay, we found different results for nitrogen additions to small vs. large plots. Small plots produced tall grass as desired (Boyer and Zedler 1998), but larger plots included areas of sedimentation and places where a native succulent outcompeted the desired grass (Boyer and Zedler 1999; Section 1.3.3). Hence, we caution those who implement restoration projects that management regimes based solely on small-scale studies will have some surprising outcomes. It is not always possible to extrapolate cause-effect patterns at the small scale to whole systems, because a subset of the ecosystem may lack predators, disturbances, invasive species, and other factors that

can shift the course of restoration. At the very least, new ideas for restoring wetlands should be evaluated in light of broad ecological knowledge of the system in question. Even then, there may be surprises. Following are four examples of events that were unpredictable from our previous knowledge of natural salt marshes.

1.3.1 Short cordgrass

When the California Department of Transportation (Caltrans) planted cordgrass (*Spartina foliosa*) in two San Diego Bay constructed marshes (Boxes 1.7 and 1.8), managers expected a tall, lush canopy to develop, capable of attracting the endangered light-footed clapper rail (*Rallus longirostris levipes*). The question was not if, but when, it would happen and which soil amendments would accelerate the process. Although plants established over much of the area where they were transplanted, the canopies failed to grow tall. Even with nitrogen enrichment, canopies were only temporarily tall (Boyer and Zedler 1998). We were surprised that nitrogen was so limiting in both the short- and long-term; we were especially surprised that belowground reserves did not build up over time (Boyer et al. *in review*).

1.3.2 Scale insect outbreak

No one predicted that a native scale insect (*Haliaspis spartina*) would break out in canopies of short cordgrass in the transplanted marshes of San Diego Bay. This insect had never been identified as a pest, and it had never been reported to reach outbreak proportions in natural marshes. Some cordgrass plants had over 1000 scale insects, and we were concerned that fertilization with nitrogen would increase their populations. Although we had considerable literature on nitrogen-plant-insect relationships to draw on, we were again surprised when nitrogen addition reduced scale insect numbers (Boyer and Zedler 1998). Hence, our work added to the scientific debate over the susceptibility of plants to insect attack — in some cases, insects find nitrogen-rich tissues more attractive; in other cases, such as ours, nitrogen enrichment strengthens plant resistance to herbivory.

1.3.3 Effects of an annual succulent on a perennial grass

Prolonged nitrogen fertilization of constructed marsh areas in San Diego Bay produced some tall cordgrass plants, but also an unexpected superabundance of tall, robust plants of a native annual pickleweed (*Salicornia bigelovii*). It seemed unbelievable that a short annual succulent could become so aggressive when fertilized with nitrogen, that cordgrass growth would be impaired, yet it happened in Caltrans' constructed marsh (Boyer and Zedler 1999). It was even more surprising that the annual was a C_3 plant (having a photosynthetic pathway that is characteristically lower in productivity) and the grass was a C_4 plant (characteristically higher in productivity).

1.3.4 Effects of algae and coots

Although algal mats are known to occur on salt marsh soils, no one predicted the massive bloom of floating algae that developed at the "Tidal Linkage" restoration site at Tijuana Estuary (Box 1.10; Section 7.5). Nor was it predicted that a flock of about 100 coots would move onto the restoration site to feed on the algae and salt marsh seedlings, simultaneously trampling many of the small plants (Section 7.4). Coots are sometimes found resting or foraging around the edges of natural salt marsh channels, but any damage to mature vegetation is not obvious.

1.4 New understanding

1.4.1 Restoration sites follow unique developmental patterns

In retrospect, it is possible to explain why surprise events (Sections 1.3.1 to 1.3.4) were not predictable from studies of natural salt marshes — in every case, the restoration process included some condition that was *not* natural. Restoration sites are not identical to situations described by secondary succession, almost by definition. Restoration is an attempt to alter the events of secondary succession by contouring sites, amending soils, planting vegetation, etc. In most cases, the intent is to accelerate succession, but, as indicated above, the result is sometimes a shift in course. Below, we draw on the surprises we encountered to update our understanding of ecosystem development in restoration settings.

The following explanations, which are really hypotheses, emphasize the feature of the restoration setting that is not shared by natural marshes. In the first instance (Section 1.3.1), the cordgrass could not grow tall because the substrate consisted of sandy dredge spoils, which could not supply or hold on to sufficient nitrogen (Langis et al. 1991). Natural marshes occur in quiet waters with fine sediments; sandy sediments are found naturally where there is more wave energy. The project imposed cordgrass onto an unnatural habitat for the species.

The second surprise (Section 1.3.2) was an insect outbreak that appears to have developed because the canopy height was too short for the scale-insect predator, a terrestrial beetle that needs tall plants as a refuge during high tide waters. Natural marshes have some tall vegetation, but the constructed marshes were mostly inundated at high tide, eliminating habitat for beetles. Had the vegetation been taller, natural checks on scale insect density would more likely have developed.

Third (Section 1.3.3), the annual succulent is a good scavenger for pulses of added nitrogen, but biweekly pulses (as in our fertilizer regimes) do not occur in natural marshes. All the marshes we have tested are N-limited, and if any species would have been predicted to outgrow cordgrass, it would have been the perennial pickleweed (*Salicornia virginica*). The annual was an unlikely pest because it does not grow tall and vigorous in nitrogen-limited natural marshes.

The algal bloom (Section 1.3.4) floated and became draped over an extensive canopy of young seedlings. Large areas with seedlings and no mature plants are unknown in our natural marsh plains. Except for annual pickleweed (*Salicornia bigelovii*), seedling recruitment is rare on the marsh plain of most southern California tidal wetlands, and this species is always interspersed among perennial species (e.g., *Batis maritima*). If a mudflat accretes enough sediment to rise to elevations where vascular plants can establish, recruitment from seed is very spotty. Most of the seaward advance of the salt marsh is accomplished by vegetative growth from the edge of the mature salt marsh canopy. Thus, natural marshes lack large mats of seedlings that could trap floating algal mats and become smothered. While algal blooms do sometimes float over the marsh at high tide, they become draped over a mature, rigid canopy, and the mats die from exposure. The coots that were attracted to this algal bloom would ordinarily not spend much time foraging in salt marsh habitats, as they avoid dense canopies of vegetation. Because the area was very open, coots were able to move into the restoration site, and they did.

The above examples differ substantially from secondary succession scenarios. Had these restoration sites been allowed to develop on their own, we would probably have seen the invasion of a few generalist species, e.g., the annual and perennial pickleweeds. In fact, in places where no plants were introduced in the first year, these were the primary colonizers, with the annual especially abundant. While the events described here are probably specific to southern California, there is a general message: Restoration sites

should not be expected to mimic secondary succession pathways when the site and the conditions imposed are very different from those of abandoned lands or naturally disturbed areas.

1.4.2 An improved conceptual framework: The restoration spectrum

1.4.2.1 A simple ecological restoration spectrum

Ecosystem development in restoration sites is less predictable than that of areas disturbed and allowed to develop naturally. Restoration sites tend to be more isolated from their natural landscape and more affected by human actions (urbanization, mining, logging, agriculture, etc.). I believe the major constraints that limit the development of restoration sites are the degree of degradation (the less degraded, the more predictable) and the degree of effort that goes into restoration (more effort should lead to greater predictability). *Predictability* here refers to ecosystem development over time — what path it takes and how closely the outcome matches a reference site. A sample indicator of ecosystem development might be the ratio of exotic:native species or their respective cover. One can predict that the proportion of plant cover that is exotic will gradually decrease over time, but it is harder to predict if — or when — it will ever be as low as in a natural ecosystem, or which species will be the most problematic, or in what sequence they might dominate the restoration site. *Degradation* involves damages to the site as well as the region, so this is a complicated variable. *Effort* involves modifications to topography, hydrology, soils, vegetation, and/or animals; it, too, is complicated. To begin, I present a simple conceptual framework (Table 1.1), supported by observations from a variety of restoration contexts. A complete conceptual framework would include these details, plus other variables, such as the taxa that are expected to occupy the restoration site or that are assessed in deciding if the site has met its objectives. A more complex model (Table 1.2) allows individual projects to be compared, ultimately leading to improved restoration theory (Zedler 1999).

1.4.2.2 Background

The book *An International Perspective on Wetland Rehabilitation* includes descriptions of 17 restoration sites, which range from the extensively bombed and napalmed Mekong Delta to mere tire tracks in the tundra (Streever 1999). Each of these sites is a candidate for ecological restoration, involving various attempts to put back some of what was present

Table 1.1 A simple restoration spectrum suggesting how differences in the degree of degradation of the site and the degree of effort put into restoration will affect the outcomes of ecosystem development. The "target" is held constant: namely, structural and functional equivalency with natural ecosystems of the same type. By comparing projects within cells, greater predictability of outcomes might be achieved.

Degree of effort → **Degree of degradation** ↓	**Little** effort	**Intensive** effort
Major degradation	Little effort is expended to solve major problems. **Prediction**: target not likely to be achieved.	Intensive effort will be needed. **Prediction**: target more likely to be achieved.
Minor degradation	A site with minor problems should require less effort. **Prediction**: hitting the target depends on taking the *right* action(s).	Few projects would expend extensive effort to solve minor problems. **Prediction:** those that do would be most likely to hit the target.

Table 1.2 The Restoration Spectrum. Restoration contexts form a spectrum, such that sites can be arranged by degree of degradation (of the landscape and the site) and by effort put into their recovery. Different actions (i, j, k) can be taken, and they may involve one or more ecosystem components: H = hydrology; S = substrate including microbes; V = vegetation including planting or weeding; F = faunal introduction or removal. Examples from San Diego Bay (A,B,C) fit in three different locations within the spectrum (from Zedler 1999, The ecological restoration spectrum, pages 301-318 *in* W. Streever, editor. *An International Perspective on Wetland Rehabilitation*, with kind permission from Kluwer Academic Publishers); see text for description of outcomes.

Degree of effort → **Degree of degradation** ↓	**1 action** (H_i, S_i, V_i, or F_i)	**>1 action in 1 component** (e.g., H_{ijk})	**1 action, >1 component** (e.g., H_I+S_J)	**>1 action, >1 component** (e.g., S_{ijk} + V_{ijk})	**>1 action, all components** ($H_{ijk}+S_{ijk}+V_{ijk}+F_{ijk}$)
Major (large area, intensive damage)	Limited effort, not likely to meet goal				Numerous actions will likely be needed
Moderate-Major		**B:** Wetland and channels excavated from fill, connected to tidal source (H_{ij}) to attract fish		**C:** Marsh excavated from fill, connected to tidal source (H_{ij}), fertilized (S_i), planted & weeded (V_{ij})	
Moderate					*
Minor-Moderate		**A:** Endangered plant reintroduced (V_i) to extant wetland		*	*
Minor (small area, minor damage)	One action might suffice		*	*	*Examples unlikely in this cell

before recent disruptions. But each situation is unique. Individual restoration sites differ in size, cause of degradation, physical impacts, biological impacts, and proposed restoration measures. With so many variables, it is naïve to think that a simple model can predict either the pathway or the specific outcome of any restoration project. A more complex framework is needed, which acknowledges both the variety of contexts as well as the variety of measures used to speed progress or direct the course of restoration.

The 17 case studies in Streever's book differ greatly in the degree of site degradation and the degree of effort put into restoration. If we search for generalities where these two attributes are similar, we may gain greater predictability. That is, we should evaluate restoration outcomes as a function of degradation and restoration effort. It might be predicted that sites with minimal degradation and major effort would follow relatively smooth pathways and achieve targets (Table 1.1). An example is Curtis Prairie at the University of Wisconsin-Madison Arboretum. This early restoration site was a little-degraded horse pasture in 1934. After much coddling over 65 years, it sustains about 170 species of native plants and many native animals. In comparison, sites with major

degradation and little corrective action are less predictable. While we can expect them to develop into something other than the kind of system that was present before disturbance, we do not know which opportunists will invade and which will eventually dominate. Surprises are likely.

1.4.2.3 A more detailed conceptual framework

Both degradation and effort are more complicated than depicted in Table 1.1. Degradation is at least a two-dimensional factor that includes damage to the site and to the region. Our sites at San Diego Bay are considered to be highly degraded in both categories (NRC 1992). Effort is also multi-dimensional, as restorationists can manipulate topography, hydrology, soils, vegetation, and fauna, each in multiple ways.

An elaborated framework is the "restoration spectrum" (Table 1.2, Zedler 1999), which includes several degrees of degradation and effort. Placed within the spectrum are the results of three San Diego Bay restoration efforts (A,B,C), each with a unique position. Their different starting conditions may explain why the outcomes differed — the endangered plant (*Cordylanthus maritimus* ssp. *maritimus*) population was restored to a marsh that once supported it ("A" hit the target), but the endangered clapper rail (*Rallus longirostris levipes*) was not attracted to a marsh that was constructed from dredge spoil ("C" missed the target). In addition, several attributes of the clapper rail habitat were monitored for a decade, and each followed unique pathways and had unique outcomes. Soil organic matter leveled off at 75% of that in the reference wetland; soil TKN rose slowly, predicted to match the reference site 40 years after construction, and the number of tall cordgrass stems declined (Zedler and Callaway 1999). In example "B," habitat was created for fishes, in order to support feeding by the California least tern (*Sterna antillarum browni*). In this case, fishes were more abundant and as species-rich as in natural channels, although the type of channel constructed (large, deep) may not support all fish functions (e.g., reproduction that occurs in small, shallow creeks). Nevertheless, the specific target was hit, probably because most fishes and their foods are highly mobile, and the key factor (deep-water tidal habitat connected to natural channels) was provided.

The restoration context likely affects site potential, with less-degraded sites able to hit targets in a short time period with few actions, while more degraded sites (e.g., a marsh excavated from dredge spoil) may never hit the target even if many actions are taken. Furthermore, individual ecosystem components may react differently over time, some coming much closer to targets than others, and with trajectories following different pathways or the same pathways at different rates. This two-dimensional spectrum may still be too simple; additional axes might explain more of the variance in restoration outcomes. However, attributes such as the degree of isolation of the site, its connectivity with natural ecosystems, and the restorability of large-scale disturbance regimes could be folded into the degradation axis. Likewise, the threat of exotic species and the degree to which hydrology can be restored are of interest and may warrant individual axes. For example, some exotic species may be controllable, while others are not, making it difficult to simplify the "threat of exotics." *Avicennia marina* was eradicated from Mission Bay Marsh in San Diego in the 1980s through continual pulling of saplings and seedlings until none remained. In contrast, annual grasses in the high intertidal marsh have large seed banks and ample seed sources in the nearby upland; widespread, abundant species, such as *Parapholis incurva* and *Polypogon monspeliensis,* will never be eradicated. Similarly, in urban areas, the hydrology of salt marshes is permanently altered by increased freshwater runoff from rooftops, streets, and sidewalks, by contaminants that are washed from streets and lawns, and by insufficient tidal water where culverts or tide gates separate the site from the ocean. Restoration of natural inundation and salinity regimes is nearly impossible; it certainly cannot be done without major technological installations and continual maintenance.

The plight of Ballona Wetland (Box 1.1), just north of the Los Angeles Airport, is a stellar example, detailed in the documentary film, *The Last Stand* (Echo Mountain Productions 1999). Most of the natural marsh was dredged to become Marina del Rey, a pleasure boat-oriented residential community. The tiny salt marsh remnant has its tidal flows reduced by small culverts that open into a concrete flood control channel, which in turn opens to the ocean. The current private landowners closed the tide gates, allowing the marsh to become very dry and hypersaline. Upstream, urban runoff has increased the freshwater inflows, bringing brackish species further down the drainage channel that dissects the salt marsh. A major street runs diagonally across the marsh, further impairing the potential for restoring tidal flows — not only does it block surface flows, but also its low elevation precludes planning for full tidal flushing, as the road would be flooded at extreme high water.

The problems of exotics and altered hydrology are further treated in Chapter 7. The challenge in developing restoration theory is to accumulate enough comparable data to quantify the influence of each constraint on restoration outcomes. We need to know which factors explain significant variance in pathways and outcomes; those can form the basis of future versions of the restoration spectrum.

Although there are many variables that need to be understood in order to predict the outcomes of restoration efforts, degradation and effort are so different among projects that these factors need to be taken into account when comparing pathways and rates of progress. In other words, we should seek common pathways and performance curves *within* cells of the restoration spectrum, and not be surprised if we find major differences *between* cells of the spectrum. Bradshaw's (1984, 1985, 1997) linear, all-purpose model is far from adequate for describing how restoration sites will progress and how ecosystem structural and functional attributes will change over time. The restoration spectrum is offered as an organizing tool for making comparisons and developing predictability. But an adequate theory of site development will not be achievable until we have more projects being evaluated in greater detail and over longer periods of time. Restoration ecologists need the resources and time to understand the complexity of habitat development in the face of very different types of sites and with very different efforts.

1.5 Adaptive restoration

The term adaptive *restoration* is introduced to describe an experimental approach to adaptive management. Adaptive *management* is a broad concept that acknowledges our insufficient information base for decision-making. It involves an iterative approach, with scientists providing information and suggestions, recommendations incorporated into management, results followed with further research, and so on. The managers take advantage of the research findings, and the researchers take advantage of the management setting to answer questions about cause-effect relationships. Our work at San Diego Bay is an example of adaptive management (see Box 6.1) — we used the sites that Caltrans constructed for endangered species (Boxes 1.7 and 1.8) as places to do research. But the sites were not designed for research; the field experiments came later, as add-ons. As a result, we had to fit the experiments to the site, rather than design the site to fit experimental needs. Adaptive restoration involves this additional step — making the restoration an experiment with alternative actions tested in replicate. This approach makes it possible to evaluate cause-effect relationships with rigor and to speed the learning process.

In the absence of highly predictable outcomes, it becomes more and more important to capitalize on the research opportunities afforded by restoration sites. Wherever possible, restoration projects should incorporate experimentation into the design, focusing on the

most important questions about how to make the site achieve its goals. While the process of adaptive restoration might not ensure the desired outcome, it will suggest corrective measures and/or lead to better restoration with the next effort in that type of ecosystem.

Tijuana Estuary provides two examples of adaptive restoration. At the Tidal Linkage (Box 1.10), we needed to know how many and which species should be planted on the marsh plain. Most of our previous restoration research had considered cordgrass at the marsh-creek edge. We had considerable information on the historical and recent composition of marsh plain vegetation, but we were unsure about: (1) the species that would recruit on their own, and hence would not need to be planted; (2) the ones that would crowd out other species, and hence should be planted later, if at all; and (3) the combinations that would be persistent in a newly excavated area. The restoration site was large enough to accommodate 87 2 × 2-m plots, and our experimental design was incorporated into the plans for construction. We asked how each of the eight species and how randomly drawn three- and six-species assemblages would persist, how much nitrogen they would accumulate above and below ground, and how their canopy architecture would develop over time. The National Science Foundation funded the analysis of experimental outcomes, including an ambitious greenhouse experiment that replicated every species assemblage that was grown in the field (Chapter 4).

A second, much larger, experiment began in fall 1999 with construction of the Model Marsh at Tijuana Estuary (Box 1.11). Although designed in the late 1980s, this restoration site had to undergo major planning and fund-raising before construction, to ensure that the excavation of flood-borne sediments would not be reversed by new sedimentation events. Because the watershed of Tijuana Estuary is largely outside local governmental control (3/4 of the ~6800 ha are in Mexico), flash floods and sewage pump station failures in Mexico can release sudden, large flows that threaten to bring new sediments to any excavation site near the border. The State Coastal Conservancy undertook a major watershed planning project to design flood retention basins in nearby Goat Canyon; once built, these will protect the 8-ha marsh from such sedimentation.

At this site, we asked the practical question, how important is it to include tidal creeks in salt marsh designs? At the same time, we asked the more scientific question, how are ecosystem structure and function linked to topographic heterogeneity? Hence, this experiment will not only determine the benefit-to-cost ratio of adding channels and creeks to future excavations, it will also advance scientific knowledge. In all, 200 ha are slated for sediment removal in a modular program that is expected to take decades and millions of dollars. Planning for future modules will begin once funds become available, incorporating results from the Model Marsh. The experiment is also linked to restoration work planned for nearby San Dieguito Lagoon (Box 1.3), where Southern California Edison is required to restore 60.1 ha of salt marsh and channel habitat, much of which will have to be excavated to remove sediment plumes, just as at Tijuana Estuary. Additional experiments on the soil amendments, species, and transplant spacing will be incorporated into the site after it is excavated. A long list of response variables, including both structural and functional attributes, will be assessed at least for the short term. Long-term assessment will, of course, depend on funding.

Adaptive restoration shows considerable promise for answering critical questions about how to achieve predictable outcomes in tidal marsh excavations. With an understanding of what happens and why it happens, we can prescribe corrective measures for future projects and recommend the most cost-effective excavation and planting regimes. Finally, we will accumulate additional case studies that will add to the San Diego Bay experience and help develop the broader restoration spectrum model.

1.6 Restoration in coastal wetlands

The special attributes of coastal wetlands and salt marshes are treated in reference books by Chapman (1960, 1977), Barnes (1977), Adam (1990), and Allen and Pye (1992). Here we discuss the features that constrain restoration attempts, emphasizing the salt marsh as a portion of the wetland habitat complex. Our study sites in southern California also have unique features that affect restorability.

1.6.1 Challenges in coastal wetlands

The features that make natural coastal wetlands unique are the same features that create restoration challenges: Their physicochemical environment is complex; they are biologically diverse, and they are vulnerable to changing sea levels. The complex physiography of coastal wetlands is difficult to create or restore, especially where prior landforms have been obliterated through filling. In Chapters 2 and 5, we describe the necessity of incorporating dense networks of tidal creeks into constructed salt marshes. To date, the most we have seen built are deep, wide channels, which are most easily formed by a bulldozer. Backhoes, Ditch Witches, and hand-operated equipment may be needed for constructing the necessary tidal creeks. Tijuana Estuary's Model Marsh (Box 1.11) will be one of a few restoration sites, if not the first, to have complex tidal creek networks excavated in replicate.

The physiography in turn affects soil chemistry, and salt marsh vegetation is tightly linked to inundation and salinity regimes. Duplicating the natural salinity and moisture regimes is also difficult. Chapter 3 indicates how little we know of the causes of existing patterns in salt marshes, making it difficult to design marshes that will have known species distributions and canopy qualities. Very often, the plants occur today in conditions that do not support seedling recruitment. In restoration, the trick is to create conditions where the species can establish, which may be quite different from the conditions after the same species has grown to maturity. For example, sediments might have raised elevations beyond the tidal range that facilitates seed germination and growth. Recreating establishment history is thus a real challenge.

Perhaps the biggest challenge in restoring habitat for animals is their species diversity, the most numerous being insects and other arthropods. Besides the many species we know about, more are likely to be discovered among the minute fauna, such as nematodes, that inhabit marsh soils. A few invertebrates are considered threatened with extinction (the wandering skipper, *Panoquino panoquinoides*, and several species of tiger beetle, *Cicindela* spp.) in southern California. Larvae of the skipper are limited in their feeding to one species of salt marsh grass, *Distichlis spicata*; the tiger beetles have preferences for different substrate textures and moisture regimes, but restoration measures have not been attempted for either type of insect. Fishes are fewer in species and seem to respond rapidly to newly available habitat. Because the hydrology and water quality of our wetlands are both so heavily modified, it may be that the most sensitive aquatic species have long since been extirpated. Birds are the most highly valued, by birdwatchers and the general public, and we know that it is a challenge to restore habitat for endangered birds — not only do they require large areas of coastal wetland habitat (no longer available in our region), but some have very specific requirements for habitat quality (e.g., the light-footed clapper rail). We emphasize fishes and macro-invertebrates in this book (Chapter 5), as we have a better understanding of how to restore habitat for some of these species than for micro-invertebrates and birds.

In most coastal wetlands, the support of food chains leading to harvestable fish and shellfish is the most important restoration goal. Restoring other ecosystem functions, such as water quality improvement and hydrologic functions (flood abatement, shoreline

anchoring) is more a concern of regional planners. Unlike freshwater wetlands, salt marshes do not improve the quality of drinking water, but they can reduce sedimentation and eutrophication of coastal waters. In Louisiana, there is great concern about wetland loss due to subsidence, a problem that can only be solved by increasing the sediment-trapping and peat-building functions of remaining wetlands. The magnitude of the problem (losses of 66 km^2/yr for the period 1983-90; Sanzone and McElroy 1998) and the cost of engineering the Mississippi River to deposit sediments more widely make this the nation's greatest coastal wetland restoration challenge. Various U.S. cities employ salt marsh planting schemes to anchor shorelines. Such practices are well known in China and The Netherlands, where the U.S. Atlantic Coast *Spartina alterniflora* has been deliberately introduced to trap sediments and expand land area (Mitsch and Jørgensen 1989).

Rising sea level presents a long-term problem for coastal wetlands, especially where salt marshes abut cities. Accelerated rates of sea level rise accompany global warming, and the lower edges of the salt marsh will gradually succumb to excess inundation, and the entire marsh will need to move inland to compensate. Many coastal marshes will bump into berms or sea walls and hence be squeezed into narrow strips or obliterated. Even where gently sloping topography will allow the inland shift of salt marshes, it is not clear that all species can expand their distributions rapidly enough to keep pace with increasing inundation and sustain biodiversity. As the upper marsh moves inland, halophytes will have to compete with upland vegetation, which will only occasionally be at a disadvantage due to salt deposition from storm tides. An occasional inundation with seawater will probably not kill upland perennials, but more frequent wettings will favor halophytes. A few species that can expand vegetatively may dominate the upper marshes of the future. While we do not deal with this issue directly, it should not be forgotten in planning salt marsh restorations. For the salt marsh to be able to respond to rising sea level, there must be a broad wetland-upland transition zone, with very gentle slopes, as are found in most natural wetlands (see Box 2.3).

1.6.2 Challenges in southern California

Salt marshes in southern California share many of the attributes of coastal wetlands elsewhere in the world, but there are peculiarities (Zedler 1982). They occur on the "leading edge of the continent," which has a narrow continental shelf, a nearby mountain range, and small coastal watersheds. The marshes are thus small and isolated from one another. The region has 25 to 30 coastal salt marshes in a strip that is 160 km long. A comparable distance along the coast of Georgia (similar latitude) has 100 times the area of wetland.

The region's Mediterranean-type climate leads to year-round hypersalinity (soils generally >40 ppt), very seasonal freshwater inflows (usually between November and March, with unpredictable timing and amount), and high interannual variability. Tijuana Estuary did not experience flooding in the 15 years between 1963 and 1978, but in the next 15 years there were four major floods (1978, 1980, 1983, 1993), followed immediately by the floods of 1995 and 1998 and the high sea levels associated with the 1998 El Niño Southern Oscillation (ENSO). What lies ahead cannot be predicted. Recurrent ENSO events add additional variability, with sea level anomalies of 30 to 60 cm (high water marks above the highest predicted tides).

Southern California salt marshes belong to a biogeographic region that is rather limited. From Point Conception (northwest of Santa Barbara) to Baja California Norte, the coast of the Southern California Bight has somewhat warmer temperatures than central and northern California. The number of salt marshes in this region is small, and their cumulative area is not that great. The largest remaining sites are in Mexico, where only a few studies have been done, e.g., in San Quintin Bay (Box 1.12, Neuenschwander et al.

1979, Zedler et al. 1999) and Estero de Punta Banda near Ensenada (Ibarra-Obando and Poumian-Tapia 1992). Unfortunately, no censuses of habitat area include marshes in both Mexico and the U.S., and estimates of the areas of U.S. marshes are out of date. The binational jurisdiction of both the regional wetland resource and the watershed of the best-known site (Tijuana Estuary) is a unique quality that has direct bearing on restoration. Reserves of many populations reside in Mexico (e.g., the light-footed clapper rail), but reintroductions from a foreign country are not easily accomplished.

Finally, the immense human population along the southern California coast (>16 million people) has produced one of the highest rates of habitat loss and some of the most fragmented and altered wetland remnants. The typical salt marsh has been filled at its upper limit and dredged at its lower limit. It is criss-crossed by transportation corridors, as well as sewer and power lines, receives urban runoff, and is visited daily by recreationists. One near Los Angeles (Bolsa Chica) has a checkerboard pattern of dikes from former duck-club impoundments and still-operating oil wells. Throughout southern California, filling and sedimentation have reduced tidal prisms so much that ocean inlets tend to close, increasing the extremes of temperature, oxygen, and salinity. Most of the region's salt marshes have closed to tidal flushing at one time or another, sometimes for several years. Most have lost species richness as a result. Most are managed in some way to facilitate tidal flushing.

Having a high human population is not all bad for wetlands. The dense population makes cities less pleasant surroundings than the open space along the coast. People who like open space and nature tend to find one another and, where there is a critical mass, they form citizen groups, or "friends" (Friends of Famosa Slough, Amigos de Bolsa Chica, etc.), who act as environmental watchdogs. Even though local economies are not based on wetland-supported fisheries, the public places high heritage and recreational value on wetlands. Birdwatchers lead the list of wetland protectors in southern California.

As a result of past habitat loss, many coastal wetland species have become threatened with extinction. With mandates to protect endangered species and continuing pressures to widen highways and expand public services (water, power, and sewer lines), regulatory agencies require substantial mitigation for damages to coastal wetlands. Damages to a hectare of wetland often require the restoration or creation of four hectares of wetland. Mitigation has thus become the most common impetus for restoration projects. Large sums of money are invested in excavating fill and flood-borne sediments from lagoons and marshes to increase tidal prisms and facilitate tidal flushing.

1.7 Summary and conclusion

Restoration ecology is not yet at a stage where the outcome of a specific set of actions at a specific site can be predicted. Restoration research lags behind that of secondary succession, in that there are few well-studied cases and few well-studied ecosystem types. One conceptual model (Bradshaw 1984) suggests that restoration sites develop function (e.g., nutrient supply) linearly with structure (e.g., species richness), but this basic theory does not take into account differences due to site degradation and variable efforts put into restoration. The development of restoration theory has not yet led to predictability. Diverse cases fall within a restoration spectrum, with degradation and effort as the two main axes. Predictability should emerge for situations that have similar types of degradation and effort.

Salt marshes pose special problems for restoration because: (1) they are complex in their topography, inundation, and salinity regimes; (2) they support many species of animals, some that require specific habitats and linkages with other habitats; and (3) they are subject to the vertical adjustments of coasts as they undergo subsidence and sea level

rise. Marshes in each region offer unique challenges. Those in southern California are especially challenged by the impacts of a continually growing human population.

In the face of inadequate theory and major restoration challenges, I conclude that restoration efforts should incorporate more science, including the design of the restoration site as one or more experiments that will answer the key questions for its own and future restoration efforts.

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Box 1.1 Ballona Wetland

Janelle M. West

BACKGROUND — As the last remaining major coastal wetland in Los Angeles County, Ballona Wetland has been nominated as a "wetland of international importance" in the film *The Last Stand*. Historically, Ballona Wetland was part of an expansive 800+ ha coastal wetland habitat. The construction of rail lines in the late 1800s and roadways in the early 1900s broke up this wetland into smaller, isolated parcels. Ballona Creek was channelized in 1934, and development increased in surrounding areas. The majority of remaining wetland was excavated to create Marina del Rey in the 1960s.

Today, the remnant wetland is entirely surrounded by the extensively developed metropolitan area of Los Angeles; all connectivity to natural upland habitats has been lost (Figure 1.1). The most pressing threat to this wetland is a proposed development, Playa Vista, which, if actualized, would be the largest real estate development in the U.S. in this decade (0.56 km^2; Echo Mountain Productions 1999). Located in the shadows of Hollywood, Playa Vista was initially planned to include a multi-million dollar DreamWorks studio, as well as residential communities, schools, and public services, all of which would be built adjacent to the wetland remnant.

Figure 1.1 Aerial view of Ballona Wetland. Note the channelization of waterways and the extensive development in surrounding areas.

LOCATION AND OWNERSHIP — Ballona Wetland is located on the western edge of the Los Angeles metropolitan area, just north of the Los Angeles International Airport and south of the cities of Vista Del Mar and Culver City. The majority of Ballona Wetland is privately owned, with a small portion held by the City of Los Angeles. To reach the wetland, exit Interstate 405 at Highway 90 and travel west. Exit at Culver Blvd.; you will reach the easternmost portion of the wetland after traveling 0.8 km.

SIZE — 84 ha (wetlands); 337 km^2 (watershed)

ECOLOGICAL CONDITION — In addition to the impacts of surrounding urbanization, tidal action is impaired by culverts with one-way flap gates, which allow freshwater outflow but prevent substantial seawater inflow (Boland and Zedler 1991). As a result, soils have dried and water salinity has declined. Many areas along the creeks are being overtaken by species more tolerant of brackish conditions.

Despite its precarious position in the midst of one of the world's largest cities, Ballona Wetland provides habitat for several sensitive species, including the Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*; PERL 1989) and the salt marsh wandering skipper (*Panoquino panoquinoides*). This appears to be the only significant nesting habitat for the Savannah sparrow between Long Beach and Malibu. Ballona Wetland also provides important habitat for lower-trophic level species, which provide food support for commercially important fish, such as the California halibut (*Paralichthys californicus*).

SIGNIFICANCE — Developers are facing strong opposition from many proactive individuals and resource protection groups who are fighting for the preservation of this remnant wetland. The situation has brought together concerned citizens, scientists, and resource agencies to collectively work to restore and expand the wetland, including proposals to increase tidal flushing and remove exotic vegetation.

FOR SITE-SPECIFIC INFORMATION, SEE:

Boland, J., and J. B. Zedler. 1991. The functioning of Ballona Wetland in relation to tidal flushing. Part 1: Before tidal restoration. Report prepared for National Audobon Society, Sacramento, California, USA.

Echo Mountain Productions. 1999. The last stand. Film. Echo Mountain Productions, Los Angeles, California, USA.

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Box 1.2 Anaheim Bay

Janelle M. West

BACKGROUND — Prior to 1868, Anaheim Bay was not highly disturbed. Hunting and fishing clubs, pier construction, and discovery of oil led to increased development within the bay and impacts to wetland habitats. A 183-m wide shipping channel connecting the inner and outer bays, as well as culverts and tide gates, restrict tidal flow into the inner bay; tidal action in the upper bay is muted. Periodic dredging of the shipping channel is required.

To mitigate for construction of a ~60-ha deep-water landfill in Long Beach Harbor, the Port of Long Beach was required to restore ~47 ha of wetlands. Construction of four tidal basins was completed in spring 1990. The basins are connected to the ocean via channels and culverts. Mitigation standards required that 50% of the area be subtidal, 35% low marsh, and 15% high marsh.

LOCATION AND OWNERSHIP — 33°44′N, 118°5′W. Anaheim Bay is located within the city limits of Seal Beach and Huntington Beach. To reach Anaheim Bay, exit Interstate 405 at Warner Avenue and travel west. Once you reach Pacific Coast Highway (PCH; Highway 1), turn right (north); the Bay will be on your right-hand side as you drive along PCH. The majority of Anaheim Bay is owned by the U.S. Navy, with a smaller portion held by Orange County and some private interests.

SIZE — 382 ha (wetlands), 154 km^2 (watershed)

ECOLOGICAL CONDITION — Lands adjacent to the bay are heavily urbanized; uplands include several inactive landfills. Muted tidal action in the upper bay, along with non-point source pollutants (e.g., metals and pesticides from urban runoff) have greatly impaired water quality. Despite these negative impacts, Anaheim Bay supports productive fish and invertebrate communities, including nursery habitat for some commercially important species (California halibut - *Paralichthys californicus*, spotted sand bass - *Paralabrax maculatofasciatus*, white croaker - *Genyonemus lineatus*, etc.; MEC 1995). While fish and invertebrates rapidly colonized the restoration sites in the bay, vegetation developed slowly (cover at the restoration site was not equivalent to the reference site after five years post-construction; MEC 1995). Anaheim Bay provides nesting habitat for several sensitive species, including the California least tern (*Sterna antillarum browni*), the light-footed clapper rail (*Rallus longirostrus levipes*), and Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*) (MEC 1995).

SIGNIFICANCE — The mitigation project has several important attributes. Only one reference site was used for comparison with all four tidal basins, and it was a tidal mudflat rather than a deeper basin. The choice of a reference site is critical, as this greatly affects how ecosystem development is interpreted. The standards used led to a judgment of mitigation compliance, but the system's long-term sustainability is uncertain. If sedimentation remains low, the tidal basins will not fill in; if sedimentation becomes a problem, or if culverts are not properly maintained, the system will experience reduced tidal influence. Finally, the fish habitat required was fulfilled by creating a specified area of open water. There was no requirement that fish have access to salt marsh feeding areas. Recent studies (Williams and Zedler 1999, Desmond et al. 2000, West and Zedler *in press*) indicate that both intertidal and subtidal habitats are useful for fishes. Including both of these habitat types may afford some fishes a large energetic advantage (West and Zedler in review, cf. Box 5.1).

FOR SITE-SPECIFIC INFORMATION, SEE:

Desmond, J. S., G. D. Williams, and J. B. Zedler. 2000. Fish use of tidal creek habitats in two southern California salt marshes. Ecological Engineering, **14**:233-252.

MEC Analytical Systems, Inc. 1995. Anaheim Bay biological monitoring project. Final report submitted to Port of Long Beach. MEC Analytical Systems, Inc., Carlsbad, California, USA.

West, J. M., and J. B. Zedler. *In press*. Marsh-creek connectivity: fish use of a tidal salt marsh in southern California. Estuaries.

Williams, G. D., and J. B. Zedler. 1999. Fish assemblage composition in constructed and natural tidal marshes of southern California: relative influence of channel morphology and restoration history. Estuaries **22**:702-716.

Box 1.3 San Dieguito Lagoon

Janelle M. West

BACKGROUND — San Dieguito Lagoon (SDL) was historically the largest of the coastal lagoons in San Diego County. In the past century, however, major filling activities have occurred in SDL, including construction of fairgrounds, parking lots, a shopping center, and several roadways (Figure 1.2). Only half of the original marshlands remain, and those are highly modified. Sewage treatment ponds were constructed in the western portion of the lagoon; up to 1.1 million L of sewage were discharged daily for a 34-year period. The sewage inflows resulted in the formation of a thick layer of sludge on the lagoon bottom. The construction of two dams upstream reduced freshwater inflows. The combination of these events led to mouth closure in the 1940s; the mouth washed open only during large winter storm events. It was not until 1978 that the California Department of Fish and Game initiated a lagoon enhancement project and reopened the mouth to restore tidal flushing.

LOCATION AND OWNERSHIP — SDL is located 32 km north of San Diego in the city of Del Mar. To reach SDL, exit Interstate 5 at Via de la Valle and travel west until you reach the beach. Take a left on Camino del Mar; you will see the lagoon inlet just after the turn. Approximately half of SDL is privately owned, while the remaining half is owned by several public agencies (California Department of Fish and Game, San Diego County, City of San Diego, San Diego River Park Joint Powers Authority, North County Transit District, and the 22nd District Agricultural Association).

SIZE — 104 ha (lagoon), 897 km^2 (watershed)

ECOLOGICAL CONDITION — Although it is a highly modified system, SDL provides habitat for two endangered species: California least terns (*Sterna antillarum browni*) forage

Figure 1.2 View of San Dieguito Lagoon. The Del Mar Fairgrounds are in the background; Interstate 5 is on the far right.

in the channels, and Belding's Savannah sparrows (*Passerculus sandwichensis beldingi*) nest in the pickleweed marsh. SDL is also home to numerous benthic invertebrates, shorebirds, and a wide variety of fish (dominants include topsmelt - *Atherinops affinis*, longjaw mudsucker - *Gillichthys mirabilis*, deepbody anchovy - *Anchoa compressa*, yellowfin goby - *Acanthogobius flavimanus*, arrow goby - *Clevelandia ios*, and shadow goby - *Quietula-y-cauda*; MEC 1993). Sedimentation has greatly reduced tidal action; the lagoon is flushed only intermittently, and periodic dredging will be required to maintain flushing.

SIGNIFICANCE — Southern California Edison is required to restore 60.75 ha of self-sustaining wetland as mitigation for the continued operation of the San Onofre Nuclear Generating Station. This restoration project provides an opportunity to apply the valuable information we have gained concerning southern California salt marshes and their functioning directly to the design of this project. Other projects are also proposed in other areas of the lagoon, which, when actualized, will greatly improve the health of this system.

FOR SITE-SPECIFIC INFORMATION, SEE:

MEC Analytical Systems, Inc. 1993. San Dieguito Lagoon restoration project biological baseline study March 1992-May 1993. Draft technical memorandum submitted to Southern California Edison Company. MEC Analytical Systems, Inc., Carlsbad, California, USA.

Box 1.4 Los Peñasquitos Lagoon

Janelle M. West

BACKGROUND — Los Peñasquitos Lagoon (LPL) is a salt marsh with tidal channels — a coastal wetland that formed several thousand years ago when rising sea levels drowned a former river valley and fine sediments accreted behind a coastal dune system. Today, the mouth of this lagoon frequently closes to tidal flushing because several structures restrict the volume of water that can move in and out with the tides. Flushing is impeded by a railroad, which cuts through the center of the marsh, as well as a berm over a sewer line in the southeastern portion of the lagoon. In addition, a highway bridge restricts the seasonal migration of the ocean inlet as sand moves along shore. Finally, extensive development in the watershed has substantially increased the flows of freshwater and sediment to the lagoon. The cumulative impact of all these hydrologic modifications is increased accumulation of sand and cobbles at the lagoon mouth. When closure occurs, it is necessary to clear the inlet by bulldozer.

LOCATION AND OWNERSHIP — 32°56′N, 117°15′W. LPL is located on the northwest border of the City of San Diego, just south of the City of Del Mar. To reach the lagoon, exit Interstate 5 at Carmel Valley Road and travel west for approximately 1.5 km. The majority of LPL is publicly owned by the California Department of Parks and Recreation, the State Coastal Conservancy, and the City of San Diego, with a small portion held by private interests.

SIZE — 252 ha (lagoon), 255 km^2 (watershed)

Figure 1.3 As a result of increased freshwater inputs, many brackish species, such as *Typha* sp. have invaded Los Peñasquitos Lagoon.

ECOLOGICAL CONDITION — LPL was closed to tidal flushing for several sequential years, during which time the salinity of the lagoon water and soils experienced extremes (Carpelan 1969). This prolonged period of closure is likely responsible for the loss of many salt marsh species that occur in continuously flushed coastal wetlands. Purer (1942) reported lush stands of cordgrass (*Spartina foliosa*) at the site in the 1930s; this species is no longer present. Because of the increased freshwater inflows, non-salt marsh species have invaded LPL (Figure 1.3), particularly in the upper reaches of the lagoon, which experience the lowest salinities (Noe 1999).

Despite the negative impacts LPL experiences, it still supports a diverse assemblage of fishes and invertebrates (Williams et al. 1998). Large numbers of fish-eating birds can be found at LPL. Although the vegetation has low species richness, the dominant pickleweed (*Salicornia virginica*) provides nesting habitat for the endangered Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*).

SIGNIFICANCE — LPL has an adaptive management program in place and a long-term monitoring program for water quality, fish, invertebrates, and vegetation. The LPL Foundation is the responsible management authority; this group includes citizens and representatives of local government; collectively, they play an active role in enhancing and preserving the lagoon. LPL Foundation has funded physical, chemical, and biological monitoring of the lagoon since 1991. This has allowed the Pacific Estuarine Research Laboratory (PERL) to collect data and document the many changes occurring in this system, e.g., invasion of non-salt marsh vegetation, lowered salinities, hypoxia (low dissolved oxygen levels during mouth closure events). PERL staff use this knowledge to give pertinent, practical advice on the management of this unique system.

FOR SITE-SPECIFIC INFORMATION, SEE:

Carpelan, L. H. 1969. Physical characteristics of Southern California coastal lagoons. Pages 319-334 *in* F. B Phleger, editor. Lagunas Costeras. Universidad Nacional Autónoma de México, México.

Coppock, D., editor. 1985. Los Peñasquitos Lagoon enhancement plan and program. Prepared by the Los Peñasquitos Lagoon Foundation and the State Coastal Conservancy.
Noe, G. 1999. Abiotic effects on the annual plant assemblage of southern California upper intertidal marsh: does complexity matter? Doctoral thesis. San Diego State University, San Diego, California, USA.
Purer, E. 1942. Plant ecology of the coastal salt marshes of San Diego County. Ecological Monographs **12**:82-111.
Williams, G., G. Noe, and J. Desmond. 1998. The physical, chemical, and biological monitoring of Los Peñasquitos Lagoon. Annual report for the Los Peñasquitos Lagoon Foundation. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, California, USA.

Box 1.5 Crown Point, within Mission Bay

Janelle M. West

BACKGROUND — Mission Bay has experienced substantial anthropogenic alterations in the past 150 years. In 1850, the San Diego River was diverted from San Diego Bay, so that it now flows into Mission Bay (or False Bay, as it was originally named). Today, the coastal section of the river flows within rip-rap levees, and Mission Bay is an aquatic park that meets a variety of recreational needs.

The habitats of Mission Bay include sandy bottom shallow water, expansive eelgrass beds, rocky shoreline, mudflats, and intertidal marsh. There is a Northern Wildlife Preserve in the northern portion of the bay, which encompasses 93 ha of remnant salt marsh and mudflats on Mission Bay. To mitigate for recent losses of intertidal habitat, the City of San Diego established the Crown Point mitigation site (3.6 ha; Figure 1.4), which is adjacent to the Northern Wildlife Preserve. The Crown Point site was first opened to tidal flushing in December 1995.

Figure 1.4 Crown Point mitigation site on Mission Bay. The Northern Wildlife Preserve is in the background; Mission Bay is on the far right.

LOCATION AND OWNERSHIP — 32° 47′ N, 117° 14′ W. To reach Mission Bay, exit Interstate 8 at Sports Arena Blvd., turn right (northwest). Continue for 2 to 3 km (road becomes Ingrahm Street); various public access areas are in this vicinity. To reach the Northern Wildlife Preserve, exit Interstate 5 at Grand Avenue and travel west. Turn left (south) on Morell Avenue to Crown Point Drive. The majority of Mission Bay is owned by the City of San Diego, with a portion of the Northern Wildlife Preserve owned by the University of California.

SIZE — 101 ha (wetlands), 137 km^2 (Bay watershed; San Diego River drains 1140 km^2)

ECOLOGICAL CONDITION — The Northern Wildlife Preserve supports the light-footed clapper rail (*Rallus longirostrus levipes*) and Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*). California least terns (*Sterna antillarum browni*) nest on several islands in the bay and forage in the Famosa Slough and the San Diego River channel (both in the southern portion of the bay). The bay supports one of the two largest eelgrass meadows in southern California. The eelgrass habitat in the bay and the marsh habitats in the Northern Wildlife Preserve support a variety of fish species; dominants include topsmelt (*Atherinops affinis*), California halibut (*Paralichthys californicus*), California killifish (*Fundulus parvipinnis*), and arrow gobies (*Clevelandia ios*).

Mission Bay is plagued by many water quality problems; two tributary streams carry many urban pollutants into the back bay. Poor water circulation in the back bay allows these pollutants to accumulate, often causing the area to be closed to recreation for weeks or months at a time. Sedimentation problems also affect the bay; two tributaries, Rose Creek and Tecolote Creek, contain high concentrations of fine sediments which exacerbate the silting problem already present in the bay. Dredging activities are necessary to maintain navigability within the bay.

SIGNIFICANCE — Restoration of wetlands (totaling ~16 ha) has been proposed for several areas of the bay (Wallace, Roberts, and Todd et al. 1994). The restored wetlands would be designed as water treatment aids to improve water quality in the bay. It is estimated that coliform removal efficiency in a tidal marsh in Mission Bay could approach 90% (Wallace, Roberts, and Todd et al. 1994). Mission Bay provides an excellent opportunity to restore critical wetland habitat by removing fill, as well as to examine the usefulness of wetlands as "water purifiers" in a highly urbanized setting.

FOR SITE-SPECIFIC INFORMATION, SEE:

Wallace, Roberts, and Todd, Noble Consultants, Nolte and Associates, Butler Roach Group, Economics Research Associates, Wilbur Smith Associates, and D. Antin. 1994. Mission Bay Park master plan update. Prepared for the City of San Diego. Wallace, Roberts, and Todd, San Diego, California, USA.

Box 1.6 Sweetwater Marsh, within San Diego Bay

Janelle M. West

BACKGROUND — The Sweetwater Marsh National Wildlife Refuge (SM) encompasses a patchwork of salt marsh vegetation and channels along with a variety of anthropogenic alternations (dredged flood control channel, bermed roadways, etc.). Despite these many

Figure 1.5 Sweetwater Marsh has continuously maintained its connection to San Diego Bay (background).

disturbances, SM represents the largest intact area of remaining wetland acreage on San Diego Bay (Figure 1.5). SM is always open to tidal action; there is no record of tidal closure.

LOCATION AND OWNERSHIP — 32°38′N, 117°06′W. SM is located in the southeastern corner of San Diego Bay, and lies within the cities of Chula Vista and National City. To reach SM, exit Interstate 5 at E Street. Turn right, directly into a parking lot. Personal vehicles are not allowed within SM; a shuttle service provides visitor transportation to/from the Chula Vista Nature Center, which houses displays of the flora and fauna, and maintains several nature trails. Permission from the U.S. Fish and Wildlife Service (USFWS) is needed in order to access other areas of the marsh.

SIZE — 128 ha (wetlands); 518 km^2 (watershed)

ECOLOGICAL CONDITION — The watershed is quite disturbed; it includes two reservoirs which have decreased the freshwater inflows from the upper watershed. In the lower watershed, however, urban runoff has increased substantially. A small portion of the marsh has become dominated by cattails (*Typha domingensis*), which likely established during a period of high rains and increased runoff. Although salinities have increased, the cattails remain, preventing the establishment of salt marsh vegetation. Exotic plants are also a problem in other areas of SM: *Mesembryanthemum nodiflorum* and *M. crystallinum*, *Rumex crispus*, *Chrysanthemum* spp., *Sonchus oleraceus*, and *Foeniculum vulgare* have invaded many high marsh areas.

SM supports a productive fish assemblage (dominants include the longjaw mudsucker - *Gillichthys mirabilis*, topsmelt - *Atherinops affinis*, California killifish - *Fundulus parvipinnis*, and arrow goby - *Clevelandia ios*). There are two exotics present as well (yellowfin goby - *Acanthogobius flavimanus*, and sailfin molly - *Poecilia latipinna*), both of which are aggressive species whose continued presence may stress the aquatic ecosystem (Williams et al. 1998). During high spring tides, the marsh surface at SW is used by several species, primarily *F. parvipinnis* and *G. mirabilis*, and provides a rich foraging area for these fish.

SIGNIFICANCE — Despite the degradation it has experienced, this wetland provides critical habitat for many species. SM is home to six species of endangered birds (the light-footed clapper rail - *Rallus longirostrus levipes*, the California brown pelican - *Pelicanus occidentalis*, the peregrine falcon - *Falco peregrinus*, the western snowy plover - *Charadrius*

alexandrinus nivosus, the California least tern - *Sterna antillarum browni*, and the Belding's Savannah sparrow - *Passerculus sandwichensis beldingi*) and two endangered plant species (salt marsh bird's-beak - *Cordylanthus maritimus* ssp. *maritimus*, and yerba reuma - *Frankenia palmeri*). SM is the only location in the U.S. where yerba reuma occurs.

FOR SITE-SPECIFIC INFORMATION, SEE:

West, J. M., and J. B. Zedler. *In press*. Marsh-creek connectivity: fish use of a tidal salt marsh in southern California. Estuaries.

Williams, G. D., J. S. Desmond, and J. B. Zedler. 1998. Extension of two nonindigenous fishes, *Acanthogobius flavimanus* and *Poecilia latipinna*, into San Diego Bay marsh habitats. California Fish and Game **84**:1-17.

Box 1.7 Connector Marsh, within Sweetwater Marsh

Janelle M. West

BACKGROUND — The Connector Marsh (CM) was constructed and planted with cordgrass in 1985 as mitigation for the widening of Interstate 5, and the construction of the Interstate 5/Highway 54 interchange and a new flood control channel (Figure 1.6). The

Figure 1.6 Connector Marsh mitigation site. The Interstate 5/Highway 54 interchange is in the background.

mitigation requirements included provisions to provide habitat for three endangered species. In brief, the requirements were that fish be present to provide foraging habitat for the California least tern (at least 75% of the native fish species present with at least 75% natural density for two consecutive years). A population of salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*) with five patches (20 plants in each) was to be established and increase or remain stable for three years. Lastly, tall cordgrass was to be present in seven potential home ranges of light-footed clapper rails to provide nesting habitat for seven pairs.

LOCATION AND OWNERSHIP — CM is located within SM, and adjacent to the west side of Interstate 5. Permission from the USFWS is required for entrance to the marsh.

SIZE — 4.8 ha

ECOLOGICAL CONDITION — Fish proved to be rapid colonizers of the newly constructed habitats, and constructed channels held more fish than the natural channels (Williams and Zedler 1999). Fishes are able to exploit new habitats more quickly than organisms with less mobility (i.e., benthic invertebrates). Salt marsh bird's-beak expansion was initially slow where channels surrounded a remnant marsh. Plants produced few seed capsules, and pollination was hypothesized as limiting where there was little terrestrial habitat for ground-nesting bees. Salt marsh bird's-beak seeds were then sown in areas closer to the upland (where pollinators were more likely to be abundant). The population was established and grew rapidly for three years, all of which had substantial or well-timed rainfall in spring. The goal of creating tall cordgrass habitat for clapper rails was not reached. Several problems prevented the development of tall plant canopies: the soil was too sandy, it had low soil nitrogen content, and there were scale insect outbreaks on the cordgrass leaves.

SIGNIFICANCE — This restoration project provided an opportunity to document ecosystem development over a 10-year period, producing valuable data which are being used to guide and improve future restoration efforts. PERL monitored the fish and invertebrate community from 1989 to 1996, and vegetation from 1994 to 1996.

Two of the three mitigation goals were met. Compliance was quickly reached with fish community development. During the process of attempting to establish salt marsh bird's-beak on the constructed site, we learned valuable lessons concerning an important factor for success of this species — proximity to potential pollinators. Creating tall cordgrass canopies has proved to be a complicated endeavor. The difficulty in establishing clapper rail habitat further emphasizes that it is easier to protect such critical habitats while they are functioning than to recreate them from scratch.

FOR SITE-SPECIFIC INFORMATION, SEE:

Williams, G. D., and J. B. Zedler. 1999. Fish assemblage composition in constructed and natural tidal marshes of San Diego Bay: relative influence of channel morphology and restoration history. Estuaries **22**:702-716.

Zedler, J. B. 1998. Replacing endangered species habitat: the acid test of wetland ecology. Pages 364-379 *in* P. L. Fiedler and P. M. Kareiva, editors. Conservation biology for the coming age. Chapman and Hall, New York, New York, USA.

Zedler, J. B., and J. C. Callaway. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restoration Ecology **7**:69-73.

Box 1.8 Marisma de Nación, within Sweetwater Marsh

Janelle M. West

BACKGROUND — As additional mitigation for the construction of the Interstate-5/Highway 54 interchange and the flood control channel, Marisma de Nación was excavated from fill (sandy dredge spoil from the bottom of San Diego Bay) that had been deposited north of the main channel of the Sweetwater River. The site was designed as a low marsh plain for cordgrass and was excavated down to intertidal elevations appropriate for this species. Marisma de Nación has one deep, wide sinuous channel designed to provide tidal flow from the adjacent San Diego Bay, via the Sweetwater River (Figure 1.7). Marisma de Nación was opened to tidal flushing in 1990. Experimental plantings of cordgrass (*spartina foliosa*) were established just prior to opening, in order to test the ability of various soil amendments to support tall cordgrass (Gibson et al. 1994). Few native plants invaded in the first year, perhaps because their seeds had been dispersed prior to opening the site to tidal flows. The site was planted with cordgrass and other species in 1991. Large areas of pickleweed seedlings (*Salicornia virginica* and *S. bigelovii*) also established on their own in 1991.

LOCATION AND OWNERSHIP — Marisma de Nación is located within SM, just south of the flood control channel and west of the Connector Marsh. Permission from USFWS is required for entrance to Marisma de Nación.

SIZE — 6.9 ha

Figure 1.7 The sinuous channel at Marisma de Nación was designed to improve tidal flow from San Diego Bay.

ECOLOGICAL CONDITION — Although the sinuous channel was designed to be deep and wide to deliver substantial volumes of tidal water, much of the seawater flows in a sheet over the marsh, instead of rising gradually along the channel's path. The result has been the erosion of the bayward fourth of the marsh and mortality of the plantings where the elevation has become too low. Whereas many constructed channels would fill in with sediment, the limited sediment in this bay's tidal water limits accretion. Some sediments have accumulated in the inland portion of the marsh, however.

After the site was opened to tidal flushing, soils were extremely hypersaline (over 80 ppt). Although tidal action leached some salts from the marsh plain, the site was still hypersaline at the time of planting (~50 ppt). Soil salinities in the high marsh of Marisma de Nación were twice as high as those noted in natural areas, especially in unvegetated areas (Boyer et al. 1996).

Marisma de Nación has consistently supported high densities of fishes and invertebrates (Boyer et al. 1996). At least one light-footed clapper rail was sighted in the marsh on two occasions during the long-term study, but none were known to nest in the cordgrass. To reduce negative impacts on any potentially nesting rails, study of Marisma de Nación was restricted to minimal monitoring. A few small patches of tall cordgrass developed adjacent to the tidal channel at the inland portion of the marsh, where wind-blown floating debris commonly accumulated. Perhaps the frequent supply of extra nutrients from decomposition allowed the vegetation to become more robust than that of the Connector Marsh (Box 1.7).

SIGNIFICANCE — Marisma de Nación was the region's first experimental restoration site for cordgrass establishment on sandy dredge spoil. Its erosion problems led to a new understanding of channel construction: namely, that overexcavation is not appropriate for bays with low sediment supply (Haltiner et al. 1997).

FOR SITE-SPECIFIC INFORMATION, SEE:

Boyer, K., G. Williams, S. Trnka, and J. Zedler. 1996. Biological monitoring of Sweetwater Marsh National Wildlife Refuge. Annual report to the California Department of Transportation. Pacific Estuarine Research Lab, San Diego State University, San Diego, California, USA.

Gibson, K. D., J. B. Zedler, and R. Langis. 1994. Limited response of cordgrass (*Spartina foliosa*) to soil amendments in constructed salt marshes. Ecological Applications **4**:757-767.

Haltiner, J., J. B. Zedler, K. E. Boyer, G. D. Williams, and J. C. Callaway. 1997. Influence of physical processes on the design, functioning and evolution of restored tidal wetlands in California. Wetlands Ecology and Management **4**:73-91.

Box 1.9 Tijuana Estuary

Janelle M. West

BACKGROUND — Tijuana Estuary was designated as a National Estuarine Research Reserve by NOAA in 1981. It is one of the least fragmented salt marshes in southern California, and it is one of the few estuaries in southern California whose tidal inlet has almost continuously kept its ocean connection (Figure 1.8). These historical features are probably responsible for the persistence of 24 sensitive species at Tijuana Estuary

Figure 1.8 A view of Tijuana Estuary, looking south. The U.S./Mexico border lies on the hills in the background.

(Zedler et al. 1992). The majority of the estuary's watershed lies within Mexico, with the city of Tijuana surrounding the Tijuana River at the U.S./Mexico border, just a few kilometers upstream of the estuary. Because of high-density human population, inadequate sewer systems, unstable slopes and soils, and agricultural runoff, Tijuana Estuary is plagued with water quality and sedimentation problems. In recent decades the estuary has experienced sewage-contaminated inflows from the city of Tijuana, along with irrigation runoff from the adjacent urban and agricultural lands in the U.S. The sewage flows have carried various contaminants into the estuary. The irrigation runoff has lowered salinities and allowed exotic plants to establish with the upper salt marsh. Several catastrophic floods in the past 20 years have deposited large volumes of sediment into the estuary channels and salt marsh areas. Salt marsh plants have been partially buried, but a few aggressive species can grow through 10 to 20 cm of sediment. Salt marsh elevations are thus unnaturally high, and tidal inundation is reduced. Most of the salt marsh area south of the ocean inlet experiences no regular tidal flushing.

LOCATION AND OWNERSHIP — 32°34′N, 117°7′W. Tijuana Estuary lies on the U.S./Mexico Border between the cities of Imperial Beach, California, and Tijuana, Mexico. To reach the estuary from the U.S. side, exit Interstate 5 at Coronado Avenue and head west approximately 4 km. Turn left on 3rd Avenue. The visitor center will be on the right-hand side. The majority of Tijuana Estuary is owned by public entities (City of San Diego, County of San Diego, U.S. Navy, U.S. Fish and Wildlife Service, and the California Department of Parks and Recreation), with a small portion held by private landowners.

SIZE — 712 ha (wetland habitats; 1000 ha National Estuarine Research Reserve), 4420 km^2 (watershed)

ECOLOGICAL CONDITION — A period of prolonged heavy sewage flows in 1986 and 1987 resulted in a shift in the fish community from dominance by topsmelt (a pelagic species) to dominance by the arrow goby (a small burrowing species; Nordby and Zedler 1991). The bivalve community also showed a substantial response to this event, as it became dominated by smaller individuals of shorter-lived species (Nordby and Zedler 1991).

Tijuana Estuary supports San Diego County's only natural population of the endangered salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*; Parsons and Zedler 1997). The edges of tidal creeks are dominated by cordgrass, and the marsh plain is dominated by pickleweed, which provides nesting habitat for the endangered clapper rail (*Rallus longirostrus levipes*) and the state-listed Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*). The estuary provides a stopping point for many other birds along their migration route; the endangered California least tern (*Sterna antillarum browni*) and the threatened Western snowy plover (*Charadrius alexandrinus nivosus*) forage in the channels and nest in the adjacent sand dunes. Extensive mudflats also provide feeding habitat for many shorebirds.

SIGNIFICANCE — Tijuana Estuary has a long-term monitoring program for water quality, fish, invertebrates, and vegetation that has been in place since 1986. This has allowed PERL to collect data and document community responses to the many changes occurring in this system due to varying conditions (increased freshwater flows and sedimentation, El Niño events, sewage inflows, etc.).

Tijuana Estuary is a prime candidate for restoration. In addition to the Tidal Linkage and Model Marsh projects (see Box 1.10 and Box 1.11), a 200-ha restoration is planned for the south arm of the estuary. Long-term data are directly used to design and implement on-site restoration projects.

FOR SITE-SPECIFIC INFORMATION, SEE:

Nordby, C. S., and J. B. Zedler. 1991. Responses of fish and macrobenthic assemblages to hydrologic disturbances in Tijuana Estuary and Los Peñasquitos Lagoon, California. Estuaries **14**:80-93.

Parsons, L. S., and J. B. Zedler. 1997. Factors affecting reestablishment of an endangered annual plant at a California salt marsh. Ecological Applications **7**:253-267.

Williams, G. D., J. S. Desmond, and J. B. Zedler. 1998. Tijuana River National Estuarine Research Reserve: annual report on ecosystem monitoring. NOAA technical memorandum on the Tijuana River National Estuarine Research Reserve. NOAA National Ocean Service, Sanctuaries and Programs Division, Washington, D.C.

Zedler, J. B., C. S. Nordby, and B. E. Kus. 1992. The ecology of Tijuana Estuary, California: a National Estuarine Research Reserve. NOAA Office of Coastal Resource Management, Sanctuaries and Reserves Division, Washington, D.C.

Box 1.10 Tidal Linkage, within Tijuana Estuary

Janelle M. West

BACKGROUND — This small wetland habitat construction project was completed in spring 1997 (Figure 1.9). It was designed to increase the area of intertidal wetland and to improve water circulation in the northern arm of the estuary. This project also provided additional habitat for fishes and aquatic invertebrates, as well as cordgrass, marsh-plain, and high-marsh vegetation. The National Science Foundation provided funding for experimentation along one side of the tidal channel: an experiment is testing the link between plant species diversity and ecosystem functioning in this newly excavated wetland.

Figure 1.9 The Tidal Linkage restoration project at Tijuana Estuary in April 1997 before planting (A) and two years after planting in October 1999 (B).

LOCATION — The tidal linkage project is located near the Tijuana Estuary visitor center (see Box 1.9 for directions).

SIZE — 0.7 ha

ECOLOGICAL CONDITION — Physical and biological monitoring revealed that accumulation of fine sediments has converted the once hard-bottom, deep channel into a shallow, highly-organic system (Williams et al. 1998). This change limits aquatic biotic to

more tolerant, benthic fishes (killifish - *Fundulus parvipinnis* and longjaw mudsuckers - *Gilichthys mirabilis*) and deposit-feeding invertebrates (amphipods and polychaetes), and precludes filter-feeders (bivalves) and pelagic species (topsmelt - *Atherinops affinis*). The tidal linkage also provides habitat for various waterfowl and a corridor for the light-footed clapper rail (*Rallus longirostrus levipes*) to move between critical areas of cordgrass in the north arm of the estuary.

SIGNIFICANCE — The Tidal Linkage Project afforded an opportunity to test how many species are needed to develop a salt marsh in a newly excavated marsh plain. The information will be used in the next, larger restoration project, the Model Marsh. Several lessons were learned during the initial phases of construction and during the years that followed. Fine sediments excavated to connect the channel to an existing mudflat were salvaged and placed on top of the bare marsh plain to improve soil quality. Soil amendments (fine sediment and decomposed kelp) greatly accelerated canopy development. Monitoring of the physical characteristics (hydrology, water quality, etc.) concurrently with biological data (fish and invertebrate composition) provided valuable insight into ecosystem functioning.

FOR SITE-SPECIFIC INFORMATION, SEE:

Williams, G. D., J. S. Desmond, J. C. Callaway, J. Terp, and K. T. Thorbjarnarson. 1998. Report on the Oneota Tidal Linkage restoration project system 1997-1998. Submitted to the State Coastal Conservancy. Pacific Estuarine Research Lab, San Diego, California, USA.

Box 1.11 Model Marsh, within Tijuana Estuary

Janelle M. West

BACKGROUND — The restoration plan for Tijuana Estuary is driven by the need to restore tidal flushing to large habitat areas south of the ocean inlet. The Model Marsh Project is part of a 200-ha tidal restoration plan, which includes several restoration modules (Entrix et al. 1991). Questions answered and experience gained in the early modules will be used to improve later modules. The Model Marsh has been designed to answer the general question, “Is it beneficial to ecosystem development to excavate tidal creek networks prior to planting the marsh plain?” The site has been designed as a replicated experiment of six subareas: three with tidal creek networks and three without (see Figure 2.2). This will allow comparison of ecosystem development rates for these two treatments. Results will indicate the need to excavate complex creek networks in subsequent restoration modules.

LOCATION — Exit Interstate 5 at Dairy Mart Road and travel west. Continue about 4 km until you reach a kiosk just past Monument Road; park here. Travel on foot to the Model Marsh project, which is located northwest of the kiosk.

SIZE — 8 ha

ECOLOGICAL CONDITION — The Model Marsh was excavated from sediments that occur in the upland/wetland transition. Each of the six areas will have marsh plain, a main channel, and three tidal creek networks, with cordgrass (*spartina foliosa*) planted at

the lower elevations and eight to ten species of native halophytes planted further inland. Excavation was completed January 2000, and the marsh was opened to tidal flushing February 14, 2000.

SIGNIFICANCE — This experiment will directly test the importance of tidal creeks to ecosystem structure and function. PERL will evaluate many aspects of ecosystem development, including the soil characteristics, benthic and fish communities, vegetation patterns, etc. This information will help determine the value of including tidal creek networks in restoration projects.

FOR SITE-SPECIFIC INFORMATION, SEE:

Entrix, Inc., Pacific Estuarine Research Laboratory, and Philip Williams and Associates, Ltd. 1991. Tijuana Estuary tidal restoration program, Volumes 1–3. Draft environmental impact report and environmental impact statement. California Coastal Conservancy, Oakland, California, USA.

Box 1.12 San Quintín Bay

Janelle M. West

BACKGROUND — San Quintín Bay has been dubbed the “most significant salt marsh in its ecoregion” by the Endangered Habitats League. This bay is one of the largest on the Pacific coast of Baja California (the surface is about 54 km^2), and it is nearly pristine. San Quintín Bay is quite shallow (over 40% of the bay is exposed at low tide) and is covered mainly by eelgrass. The adjacent salt marshes are characterized by intricate networks of tidal channels and creeks; together they form a very complex and hetergeneous system (Figure 1.10). San Quintín Bay is also a very important economic resource; ecotourism, fishing, and a thriving aquaculture industry bring in substantial profits and employ many workers. San Quintín Bay is a top sportfishing destination in Baja.

LOCATION AND OWNERSHIP — 32°28′N, 115°58′W. San Quintín Bay is located approximately 330 km south of San Diego.

SIZE — 10,522 ha (wetlands)

ECOLOGICAL CONDITION — San Quintín Bay provides high-quality habitats, which support many species. The coastal sage scrub in the surrounding uplands support the California gnatcatcher (*Polioptila californica*), which is listed as endangered in the U.S. There is a broad transition zone between wetland and upland, which provides resting areas for wetland birds during high tides. Other terrestrial life (small mammals, snakes, lizards, and insects) also make use of this refuge. The marshes provide critical habitat for several sensitive and/or endangered species: the light-footed clapper rail (*Rallus longirostrus levipes*), least tern (*Sterna antillarum browni*), the snowy plover (*Charadrius alexandrinus nivosus*), and the black brant (*Branta bernicla nigracans*). The brant nest in the high Arctic tundra and make a spectacular migration south (a 4800 km non-stop flight) to estuaries and lagoons in Mexico and the U. S., where they spend the winter months. Their migration takes 60 to 95 hours and they lose about a third of their body weight over the course of

Figure 1.10 The marshes on San Quintín Bay are characterized by intricate networks of tidal channels and creeks.

the journey. The brant are dependent upon a steady supply of eelgrass, their main food source; San Quintín Bay's productive eelgrass beds provide an excellent foraging area.

SIGNIFICANCE — San Quintín Bay serves as a reference ecosystem for coastal wetland restoration projects in southern California, where few natural wetland habitats remain. For example, the broad wetland-upland transition zone in this natural system helps us determine what an appropriate transition zone for a restoration project should look like (i.e., size, species composition, topography, etc.).

The bay's pristine status is threatened by various potential developments, which could include hotels, golf courses, a desalination plant, large parking structures, shopping malls, and marina construction. Not only would natural habitats be lost and degraded by such development, the extensive aquaculture industry could also be adversely affected. Water quality would be impaired by such developments, potentially reducing shellfish production and negatively affecting the aquaculture industry.

chapter two

Developing a framework for restoration

Gabrielle Vivian-Smith

2.1 Introduction

Restoration aims to repair ecosystem functioning and replace biological components that have suffered losses or local extinction. A restoration model is the framework that describes in detail the desired physical, chemical, and biological attributes of the restored ecosystem. While the model should be site-specific, its attributes should also fulfill regional restoration needs. Developing a model that, when implemented, would enhance biodiversity and functioning at both local and regional scales requires an understanding of the opportunities and constraints present at the site, as well as the region. This chapter describes how these constraints and opportunities can be identified using a variety of information sources — historical, regional, and on-site. The significance of natural variation (spatial and temporal) in ecological restoration is highlighted, and ways of accommodating topographic heterogeneity in a dynamic restoration model and long-term restoration strategy are considered. Lastly, the steps involved in the development of a detailed restoration plan that takes an adaptive management approach are outlined.

2.2 Restoration goals

2.2.1 Setting goals

A broad goal applicable to most restoration or mitigation projects is to return a damaged ecosystem to a more natural condition (NRC 1992). Ideally, restoration should fulfill both regional and local goals. Regional goals and priorities can be determined by assessing both the extent of remaining wetland habitats and historical losses, not just in wetland habitat, but also in the distributions of species and levels of function, insofar as they have been documented (Zedler 1996*b*). Local restoration targets need to be set while considering the sites available for restoration (or habitat creation) and the needs of the project proponent. An agency may have funds to restore habitat for endangered species or to improve water quality, while a developer may need to mitigate damages to other wetlands to satisfy a no-net-loss policy. Thus, local targets may concern a specific habitat type, wetland function, or individual species. If regional and local goals can be matched, the project will likely have broad public support. Where suitable restoration sites are available, regulatory agencies may wish to use local mitigation funds to fulfill urgent regional needs, such as

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restoring a rare wetland type (out-of-kind mitigation), rather than creating a common wetland type to replace a lost one of the same type at the same site (in-kind mitigation). Mitigation policies, including those at the federal level, typically indicate that wetland restoration is preferable to habitat creation (e.g., from upland).

During restoration, new opportunities or constraints may develop. For example, a dredge spoil island was constructed in San Diego Bay in 1974 to mitigate damages to wetland that was excavated for a boat marina. The island was designed to include nesting habitat for an endangered marsh bird. While spoils were being pumped to the bermed island area, an endangered tern species adopted the site for nesting. Subsequent management of the site was modified to accommodate the tern's habitat needs (vegetation is mowed to sustain open sandy substrate for nesting).

The monitoring program for restoration sites should document the appearance of unexpected but desirable species. Their population growth could then be facilitated. An adaptive management approach with continual evaluation of monitoring data can improve chances for capitalizing on opportunities or correcting newly-identified problems. Targets can then be reassessed and management strategies revised when necessary during the restoration process (Zedler 1996*a*,*b*, Thom 1997).

2.2.2 *Types and value of background information*

A site-specific restoration model that complements a regional restoration plan must be built on reference information. Reference information should consist of contemporary information from local wetlands as well as historical information documenting the changes to the available site(s). This information is important as it will be used to determine which regional restoration goals are most appropriate for the site in question. Because there will be alternatives and choices to be made, the model should be viewed as a framework that can, when necessary, be modified using an adaptive management approach (Brinson and Rheinhardt 1996, Mitsch and Wilson 1996, Zedler 1996*b*, Thom 1997). A variety of sources can supply data on current hydrological and ecological conditions at a restoration site; in addition, structural and functional information on similar local wetlands and past historical details about the site will describe actual or likely changes that have occurred (White and Walker 1997).

2.2.3 *Locating reference sites*

Background information can be gained from existing local wetlands to describe ecosystem structure and functioning and contribute to the restoration model (Kentula et al. 1992, Brinson and Rheinhardt 1996). Such information will often help form the restoration model, and it may also be useful in developing performance standards for the restored site (see PERL 1990, Brinson and Rheinhardt 1996, and Chapter 6).

In heavily developed coastal regions, it is difficult to locate natural wetlands that can serve as reference sites because disturbances have so reduced the area of wetlands and altered their condition. In coastal California, NOAA (1990) estimated that 90% of the wetland area had been lost to disturbances such as dredging, dumping of dredge spoil and garbage, trampling of vegetation, off-road vehicle traffic, drainage, diking, reduced tidal flow, and altered stream flows (NRC 1992, Zedler 1996*a*, Burdick et al. 1997). Many of the remaining wetlands are isolated fragments surrounded by urban development.

In areas with dense and extensive development, the historical conditions (and therefore the restoration target) will be difficult to establish (Figure 2.1). This has been the case for coastal wetland restoration in southern California (Zedler 1996*a*) and Connecticut (Casagrande 1997) and for brackish-emergent marsh restoration in Puyallup River Estuary, Washington (Simenstad and Thom 1996). In such cases, considerable effort and extrapolation will

Figure 2.1 F and G Street marsh at Sweetwater Marsh National Wildlife Refuge is a highly degraded coastal wetland habitat.

be necessary. In the case of southern California, the historical data indicate that coastal wetlands were more influenced by tides and supported more species than is currently the case.

In areas with more extensive wetlands and less loss of wetland area, locating natural wetland sites is not as problematic. In the less developed regions of the Atlantic and Gulf of Mexico Coasts, there are extensive and intact areas of natural marsh systems that have a strong research base and can serve as models (see Mitsch and Gosselink 1993). Longer-term studies of both pristine and disturbed coastal wetlands can help describe the structure and functioning of coastal wetlands and provide a conceptual restoration model. For example, tidal creeks within the northern arm of Tijuana Estuary (Chapter 5, Figure 5.5) are the model for creeks being constructed in the southern arm's Model Marsh (Figure 2.2). Our study of the extensive, near-pristine salt marsh in San Quintín Bay, Baja California Norte, forms the basis for recommending vegetation plantings (Box 2.1). Studies of natural plant distributions and physical characteristics (e.g., elevation profiles and tidal creek distributions) are useful for designing plantings and specifying grading at coastal wetland restoration projects in the region (Box 2.1).

2.2.4 Historical records

Historical records are not only useful in locating reference sites, they can also help in the matching of regional goals with restoration site attributes. Historical information can suggest the extent of past tidal influence, the area and location of former wetlands (Dahl 1990), and former species lists. Together, such information should be integrated into a conceptual model of how the system once functioned (e.g., it was fully tidal and species-rich). Differences between this model and existing conditions (e.g., frequent closure to tidal action and large areas dominated by monotypic *Salicornia virginica*) will then suggest the restoration needs (restore the tidal prism, or the natural range of tidal heights, to increase the area of land influenced by tides).

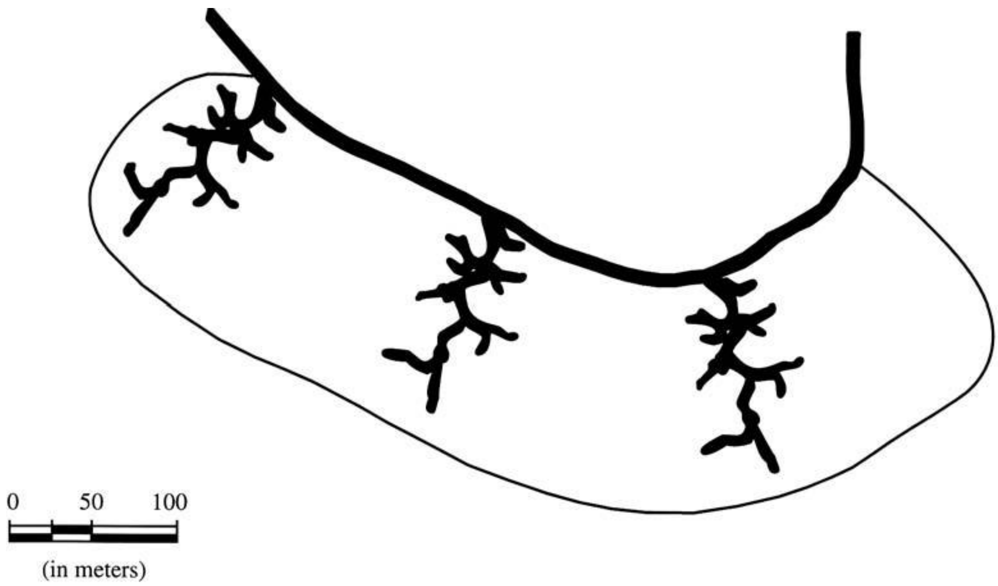

Figure 2.2 Diagram of the Model Marsh experimental restoration project at Tijuana Estuary.

Past environmental conditions of a restoration site and the surrounding area can be interpreted from a wide range of historical records, including explorers' journals; surveyors' records; herbarium collections; historical photographs; aerial photographs; old maps and oral histories (Figure 2.3, Table 2.1). Such historical documents may be overlooked in favor of more recent scientific reports, yet they often provide early records of physical conditions and organisms at the site. Detailed scientific studies of the site usually provide more recent data and may be published in scientific journals, theses, government reports, and environmental impact statements. Examples of how historical records can increase understanding of current wetland conditions and develop restoration plans concern tidal marshes of the Lower West River, Connecticut (Casagrande 1997), Long Island Sound, Connecticut (Dreyer and Niering 1995), and Tijuana Estuary in southern California (Zedler et al. 1992). While interpreting historical records, attention should be paid to factors that vary over short time scales. Maps and photographs that depict open water can document low, high, or intermediate water levels; if the topography is very gentle, the areas considered open water would differ substantially between low and high water levels.

Historical information on Tijuana Estuary includes data on sea level rise, ENSO events, rainfall, streamflow, flooding, sedimentation, and, on rare occasions, dredging of the ocean inlet. Early maps and aerial photographs depict changing land use patterns and channel morphology. Collectively, these reveal a story of many disturbance events and dramatic change at the site (Zedler et al. 1992). Williams and Swanson (1987) used the 1857 map of Tijuana Estuary to model the system as a fully tidal estuary. They measured the mouth width at 305 m and estimated the tidal prism at 1.8×10^6 m^3 and the intertidal wetland area to be 352 ha. They then calculated an 80% reduction in tidal prism between 1852 and 1986 and attributed it to sedimentation and a major retreat of the beach (Figure 2.4).

Historical information can suggest both opportunities and constraints for restoration. A map of the former wetland extent of a wetland suggests where remnant vegetation may persist (of use in providing plants or seeds for restoration efforts) or where fine sediments or hydric soils may be found (for possible salvage and reuse, if the site is not restorable). The history of constructing tide gates, dikes, or other disturbances may indicate opportunities

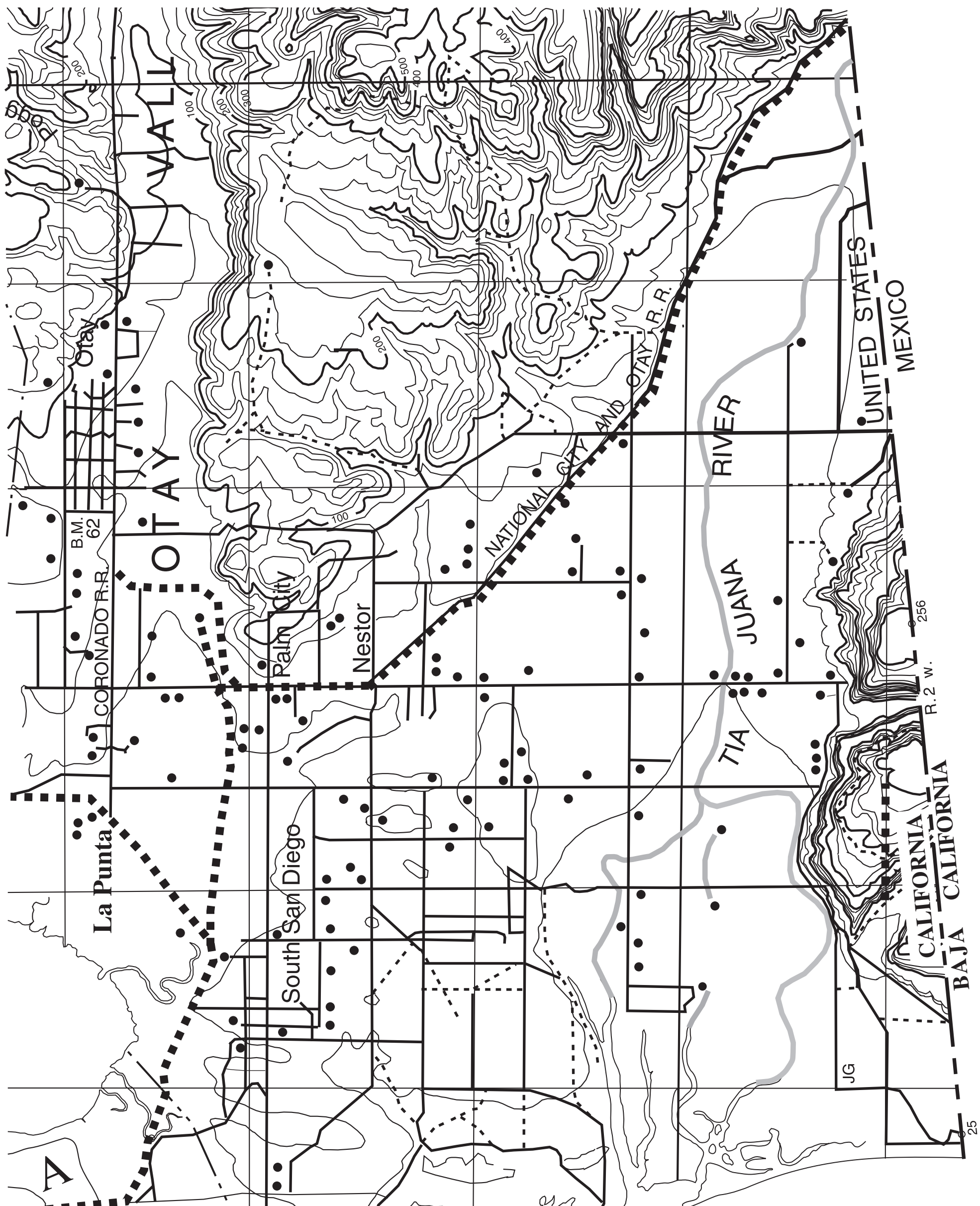

Figure 2.3 The 1904 map of Tijuana Estuary (from U.S. Coast and Geodetic Survey). Scale: 1 cm = 0.7 km.

Table 2.1 Background information sources and application in the restoration framework

Background information source	Application
Past/historical reference information	
Journals, books, papers, herbarium records, oral histories	Describe flora, fauna, and geology of site prior to development, important structures and processes
Maps, surveyors' records	Determine historical extent of marsh, channel morphology, land use
Meteorological data, streamflow records	Determine past levels of variation for rainfall
Aerial and historical photographs	Outline history of changes to marsh, channel morphology, land use
Land use and town planning records	Outline past use of marsh and surrounding area
Soil cores	Abrupt changes to soil profile in sediment characteristics indicate large sediment deposition events, dumping activities, changes in environmental conditions
Current (local reference wetlands)	
Elevation and hydrology	Indicate extent of tidal influence, patterns of tidal flow, creek density
Vegetation cover and composition	Indicate pool of local coastal wetland species and rank abundance
Water quality	Indicate variation in quality, extent of freshwater inflows experienced
Areas of high salinity	Indicate areas where vegetative colonization will be difficult
Soil properties (e.g., texture)	Indicate needs for soil amendment (e.g., nutrients, organic matter)
Remote sensing image of area	Provide measures of aerial coverage of water, vegetation, habitat areas as model for restoration site
Current (site characteristics)	
Topography and hydrology	Indicate extent of tidal influence, grading plans, drainage
Water quality	Indicate and compare variation in quality, extent of freshwater inflows
Sediment characteristics	Predict nutrient retention, drainage characteristics
Vegetation cover and composition	Indicate potential colonists, exotic species problems
Animal usage	Indicate potential colonists and permissible levels of disruption to site during construction

where minimal changes are required to reinstate tidal flows (e.g., Sinicrope et al. 1990, Brockmeyer et al. 1997, Burdick et al. 1997). Previous land uses may signal constraints for restoration; for example, former industrial activity may indicate substrate contamination (e.g., land as a waste dump, sewage ponds, or chemical disposal), while agricultural use may indicate substrate disturbance, drainage, and associated problems such as acidic discharges (White et al. 1997) or subsidence (Broome et al. 1988, Cahoon and Turner 1989, Rojstaczer and Deverel 1995, Turner and Lewis 1997).

2.3 Current site characteristics

2.3.1 Connectivity, configuration, and context within the larger landscape

Landscape ecology offers several concepts of use to restoration planning (Forman 1995). The development of the restoration model should include consideration of the position

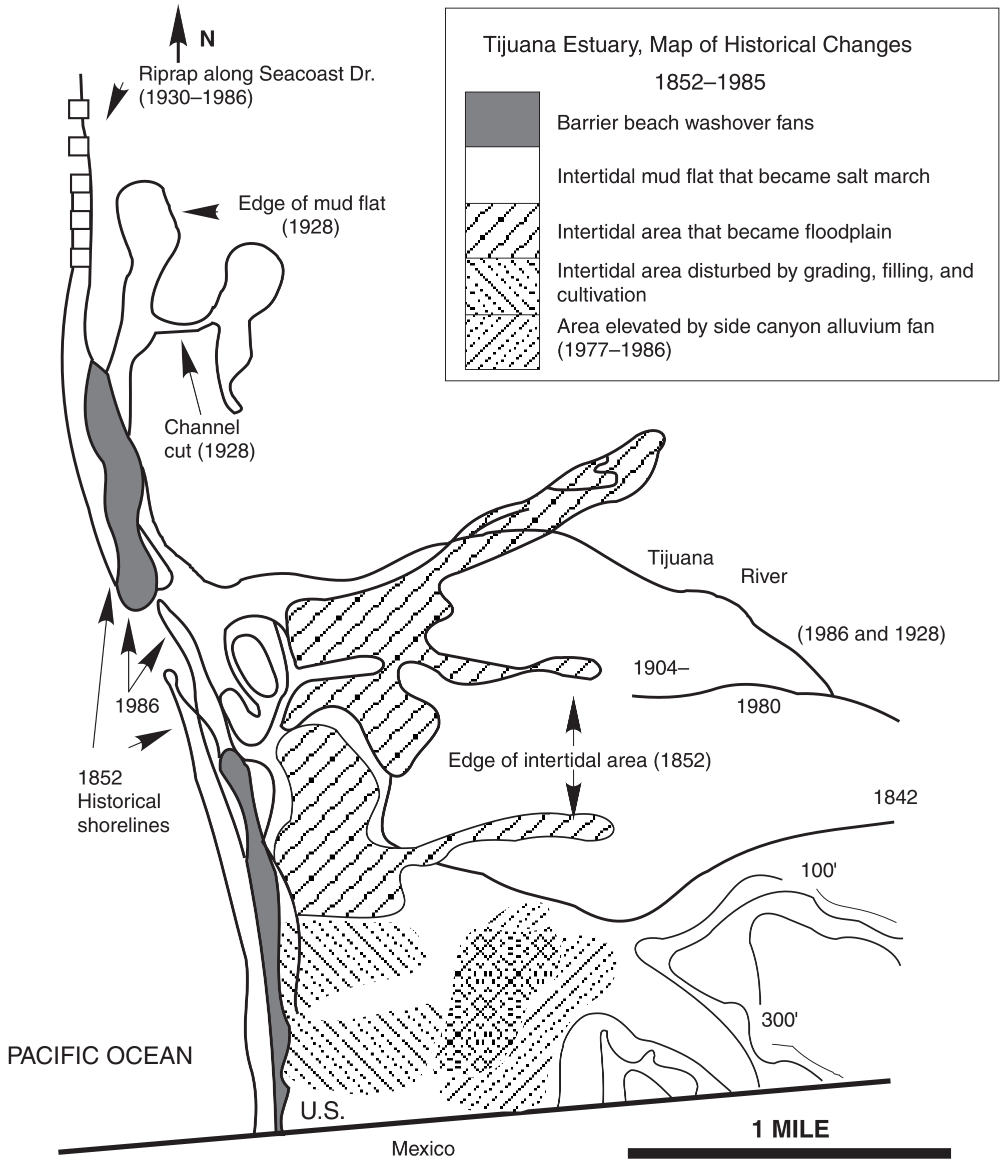

Figure 2.4 Historical changes that have reduced the tidal prism of Tijuana Estuary (modified from Williams and Swanson 1987; *Tijuana Estuary enhancement hydrologic analysis,* California State Coastal Conservancy, Oakland, CA, and 1996*a*, *Tidal wetland restoration. A scientific perspective and a southern California focus.* California Sea Grant College, La Jolla, CA, with permission).

in the landscape, connectivity of the site with other natural habitats, the relationships to adjacent land uses, proximity of natural wetlands, spatial configuration of the site (ratio of edge:interior habitat, characteristics of the wetland-upland interface, and potential area for the wetland-upland buffer).

Land use activities adjacent to or near the site may constrain the range of suitable restoration strategies. Physical barriers, such as roads and culverts, restrict water flow and the dispersal and movement of native plants and animals. Other problems may arise where steep slopes and impermeable surfaces increase freshwater runoff into the restoration site, lowering soil salinities and facilitating weed invasion (Callaway et al. 1990; Zedler

et al. 1992, Kuhn and Zedler 1997, Callaway and Zedler 1998). Increased nutrient inflows often result from wastewater spills and fertilizer use in surrounding residential areas; these can have negative consequences for the marsh by creating conditions that favor weed invasion and algal blooms (Chapter 7). Nearby soils can erode following excavation or disturbance, with excess sediment deposited in the restored marsh (Zedler 1996*a*), leading to rapid marsh infilling and unsuitable hydrology for wetland species. These problems may also be evident in local watershed management plans that reveal larger-scale sedimentation, erosion or runoff. Adjacent areas may be subject to garbage dumping, off-road vehicle use and weed invasion. Vehicle routes may attract vandals, and access may need to be restricted. High rates of vandalism suggest that restoration equipment be stored off-site and that surveillance be considered.

Connectivity with and proximity to other local wetlands are important, as nearby sources of plant and animal propagules may foster natural colonization of the restoration site and lower costs. In a North Carolina salt marsh with adjacent subtidal seagrass habitat, an intertidal salt marsh fish, *Lagodon rhomboides*, was more abundant and individuals were larger than where seagrass habitats were not nearby (Irlandi and Crawford 1997).

The spatial configuration of the site determines the edge:interior ratio and the characteristics of the buffer between the wetland and the neighboring area. Wetland remnants of small area and greater edge:interior are highly susceptible to physical disturbance and invasion by exotic species, particularly when the adjacent area is disturbed or developed. A broad buffer zone is thus needed to protect the wetland (Box 2.3). Buffer zones consisting of natural habitat, such as grassland, shrubland or woodland, can reduce disturbances due to adjacent developments (e.g., noise, traffic, artificial lighting). They also can provide a valuable refuge for birds and marsh insects during periods of flooding or extreme high tide, as well as primary habitat for small mammals that use the marsh surface during low tide (Zedler 1996*a*). Buffers should have native plantings to function more naturally (Box 2.3).

2.3.2 Site evaluation

Field evaluations of potential restoration sites and their surroundings will shape the restoration model by identifying further opportunities and constraints on restoration (Table 2.1). Field evaluation is a crucial step, as it will determine whether or not the site has cultural significance and what restoration methods are appropriate. Because wetlands were used by Native Americans, many sites have cultural significance and anthropological surveys will be needed prior to designation for restoration. For mitigation projects, the impact site should also be evaluated for opportunities to salvage soil or plant material. Specific methods for assessing sites are recommended in Chapter 6. Here we emphasize the questions of greatest importance for determining restorability.

Is there information on hydrology and vegetative cover? Recent aerial photographs, topographic maps, and remotely sensed images should be consulted (Zedler et al. 1992, Zedler 1996*a*, James 1998). Existing infrastructure should be mapped, including roads, power lines, buried pipes, and drainage systems. Site visits will help to interpret remotely sensed images and provide more detailed information on soil types, vegetation, hydrology, and animal use (see later chapters). These factors will influence the cost of restoration and the progress of the project.

What is the existing tidal regime? Site evaluation at different times during the tidal cycle will indicate the likely patterns (i.e., inundation times, frequencies, flow patterns and current strength). Hydrology of the site will also be influenced by sediment dynamics. While this is largely a function of surrounding land use, assessment of tidal flow patterns and the topography of the site and adjacent areas will suggest future sedimentation and erosion rates.

Does existing hydrology need to be improved? Site visits will augment information available on topographic maps. Detailed surveys will be needed, because the extent of tidal influence and information on elevation will shape future grading and excavation plans. Elevations that differ by 10 cm can alter the inundation and tidal flushing regime enough to affect vegetation. Instead of promoting establishment and growth, topography that is too low can cause mortality due to waterlogging, drought, stagnation, or hypersalinity. Special attention should be paid to extreme tidal events, such as rare storm tides, which bring salts inland and hinder growth by upland plants. Such areas will be more suitable for plants of the wetland-upland ecotone (Box 2.3).

Where can spoils be discharged? If excavation or dredging is indicated, alternatives for disposing of excavated material should be carefully investigated. The amount and quality of sediment to be removed (fine, coarse, presence of contaminants) and the distance transported largely dictate costs. Spoils from the Tidal Linkage (Box 1.10) were judged coarse enough for beach enhancement, while those from the Model Marsh (Box 1.11) were not, requiring that they be trucked off-site. Costs may be reduced substantially if the sediments are suitable for use elsewhere on-site or for beach rejuvenation.

What are the soil or substrate conditions at the restoration site? The soil/substrate that will be in place after grading or excavation may differ substantially from that at the surface prior to restoration. Wetland soil may be buried,which will facilitate restoration, but excavation of surface material could also reveal coarse subsoil, which may not support the desired wetland vegetation. Data on soil salinity (e.g., the presence of salt pannes or hypersaline patches), compaction, texture, moisture, and organic matter content will suggest the need for soil amendments (Chapter 3). Excavation and exposure of buried wetland sediments could create acid sulfate soils. Soil that is to be stockpiled above the water table can also become acidic following oxidation. Careful management during construction can prevent the discharge of acidic material, which could occur if rainfall leaches acids into the water, where fish might be killed. Soils that are known to be contaminated prior to restoration must be decontaminated or removed (e.g., PCB contamination at Gog-Le-Hi-Te Wetland, Simenstad and Thom 1996). If contaminants are uncovered during restoration, similar efforts will be needed. In extreme cases, the restoration project may need to be relocated. Budget planning should include funds for such unexpected problems.

What vegetation occurs at the site? Field evaluation will indicate opportunities for natural colonization and the extent of the area that will need to be planted. Remnant wetland vegetation should influence species selection, particularly where sensitive species or assemblages have persisted. The Tidal Linkage (Box 1.10), for example, was positioned to avoid an endangered plant, which appeared nearby during the planning process. Remnant habitat patches of valued species create constraints on construction disturbance; for example, nesting by endangered bird species requires that construction occur outside the breeding season. The presence of exotic plant species indicates the need for weed control and planting of natives. Any underlying factors that favor exotic species, such as excessive freshwater inputs, high nutrient levels or excessive ground disturbance, should be noted. Likewise, feral animals may prey on native species or disturb soil by burrowing or vegetation by herbivory.

2.4 Heterogeneity in coastal wetland restoration models

Natural variation (i.e., heterogeneity) occurs in both space and time. Spatial variation can be patchy or continuous; temporal variation can involve changes in abundance or rates of a particular process. Heterogeneity occurs at many spatial scales (many small patches within habitats or many habitats within a restoration site) and many temporal scales (daily, weekly, seasonal, annual, and interannual variation in rainfall, tidal inundation, and

temperature). Spatial variation influences both the functioning and diversity of many coastal wetland systems (e.g., Hartman et al. 1983, Bertness and Ellison 1987, Zhang et al. 1997). Spatial variation in factors such as salinity, moisture, redox and tidal influence is present as either continuous (gradients) or discontinuous (patchy) changes in an ecosystem's structural or functional attributes. This variation is considered to be a primary factor controlling diversity and species composition in coastal wetland communities (e.g., Pennings and Callaway 1992, Earle and Kershaw 1989).

2.4.1 Opportunities created by spatial heterogeneity

Although often perceived as a problem for characterizing coastal wetlands, natural heterogeneity is regarded as an important component of many ecosystems and is closely correlated with their biodiversity (Pickett and White 1985, Kolasa and Pickett 1991, Huston 1994, Meffe and Carroll 1994). Many coastal wetlands are highly heterogeneous systems, dissected by complex tidal creek networks and with small changes in elevation causing large differences in physical conditions (e.g., salinity levels, inundation times). Heterogeneity should be considered an integral component of a developing restoration framework by identifying and incorporating spatially dependent processes and temporal factors that control ecosystem functioning. This approach contrasts sharply with traditional agriculture, where homogeneity is the key to predictability and profit.

Biodiversity can be increased by incorporating a variety of habitats into the restoration site. This is especially important for animals that use multiple habitat types (e.g., mudflats for feeding and marsh plain for breeding or resting). Ecosystem functioning may also be enhanced by including a variety of habitats. This is particularly apparent when one habitat acts as a filter (e.g., sediment trap) or donor (e.g., nutrients) for another. Heterogeneity may also enable other spatially interactive processes such as patch dynamics and metapopulation dynamics (Wiens 1976, Wiens et al. 1993).

Restoration efforts should focus on the scales of heterogeneity that are likely to influence key ecological processes and accelerate ecosystem development (e.g., plant colonization, organic matter accumulation). At scales <5 m, one can create microtopography or patches of different soil texture or vegetation types; such patches may in turn create microsites for seedlings, provide a wider variety of environmental conditions for different species (Grubb 1977), and encourage persistence of different genotypes (Huenneke 1991). At larger scales, one can vary the distribution and numbers of habitats present (e.g., marsh plain, mudflat, creek bank, and channel) to enhance fish and invertebrate colonization. Variation in tidal creek densities and channel widths may provide breeding habitats for several fish species (Chapter 5).

Heterogeneity in restoration can act as a bet-hedging device by maximizing chances that some optimal habitat will exist at the site for intended species. Variable habitat characteristics, similar to those found in the natural habitat, may increase colonization and establishment by a greater variety of organisms (Vivian-Smith 1997). Heterogeneity may also reduce the likelihood of one species dominating the restoration site by buffering interspecific competition (Pacala and Tilman 1994), thus increasing diversity. Creating the optimal scale or type of heterogeneity in a wetland restoration is difficult and uncertain when the wetland system is not well understood. Reference sites can provide a good measure of the scale, type, and amount of natural heterogeneity, and an adaptive restoration approach can help identify the need for further modifications.

Larger-scale heterogeneity is accomplished by including two or more habitat types into the restoration plan. The habitats of coastal wetlands worldwide are subtidal habitat, tidal creeks, and channels, mudflats, mangrove forest, salt marsh, salt panne, brackish marsh and the wetland-upland transition (Box 2.5; Ferren et al. 1995; Figures 2.5, 2.6, 2.7,

Figure 2.5 A tidal channel at Tijuana Estuary.

Figure 2.6 An intertidal creek at Tijuana Estuary provides habitat for juvenile fish and many invertebrates.

and 2.8). Associated habitats include marine, riparian and upland habitats, which are linked by ecological processes, such as fluxes of materials and energy, including the transfer of biological material (e.g., organisms, propagules; see Box 2.3). The spatial arrangement and connectivity of these habitat types have an important influence on their interactions (Forman 1995).

Figure 2.7 The marsh plain at Tijuana Estuary is characterized by a diverse assemblage of species.

Figure 2.8 Salt panne habitat at Sweetwater Marsh.

Links between habitat heterogeneity and the diversity of animal and plant assemblages have been demonstrated in restored and natural coastal wetlands. Fish use many salt marsh habitats, including the marsh plain, channels, channel edges, and marsh ponds (Weinstein et al. 1980, 1997; also see Chapter 5). In coastal Virginia, low abundances of

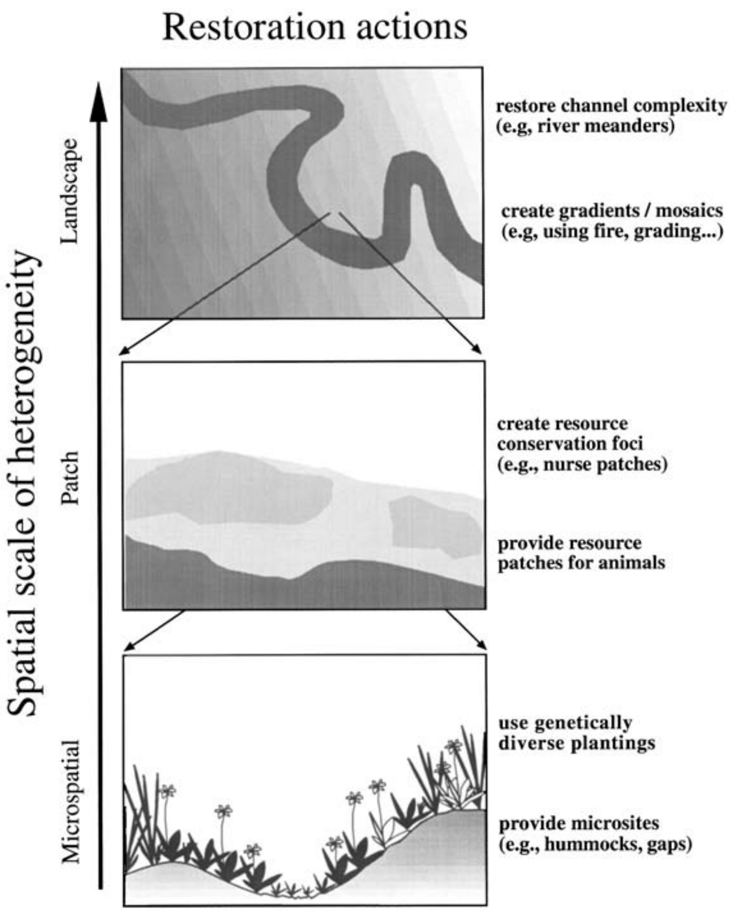

Figure 2.9 Heterogeneity and restoration considerations at different spatial scales.

shellfish and fish in a constructed marsh were attributed to insufficient morphometric heterogeneity (i.e., fewer stream rivulets) and lower levels of organic carbon (Havens et al. 1995). Benthic invertebrates are affected by salt marsh plants that offer belowground habitat (Capehart and Hackney 1989). Plant population structure also responds to microhabitat heterogeneity; the genotypic diversity of cordgrass (*Spartina patens*) appears to be maintained by the existence of marsh plain, swale, and dune microhabitats in natural East Coast salt marshes (Silander 1985). In southern California, the salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*) lives in the marsh but relies on upland areas for its pollinators. Bee pollinators nest in dry upland soils, so upland buffers with suitable bee nest sites are needed to restore this endangered annual plant (Parsons and Zedler 1997; Box 2.6). Finally, heterogeneity can enhance the persistence of species and genotypes when environmental conditions are inhospitable by providing refugia (Huenneke 1991, NRC 1992). This is particularly important for species that are sessile or unable to locate favorable microhabitats.

At each scale, spatially interactive processes may operate differently, so a restoration framework should consider ecological processes and structures at a variety of scales (Figure 2.9). Clearly, more experimental tests of the most important aspects of heterogeneity are needed within restoration sites. Other aspects of heterogeneity can then be left to develop naturally.

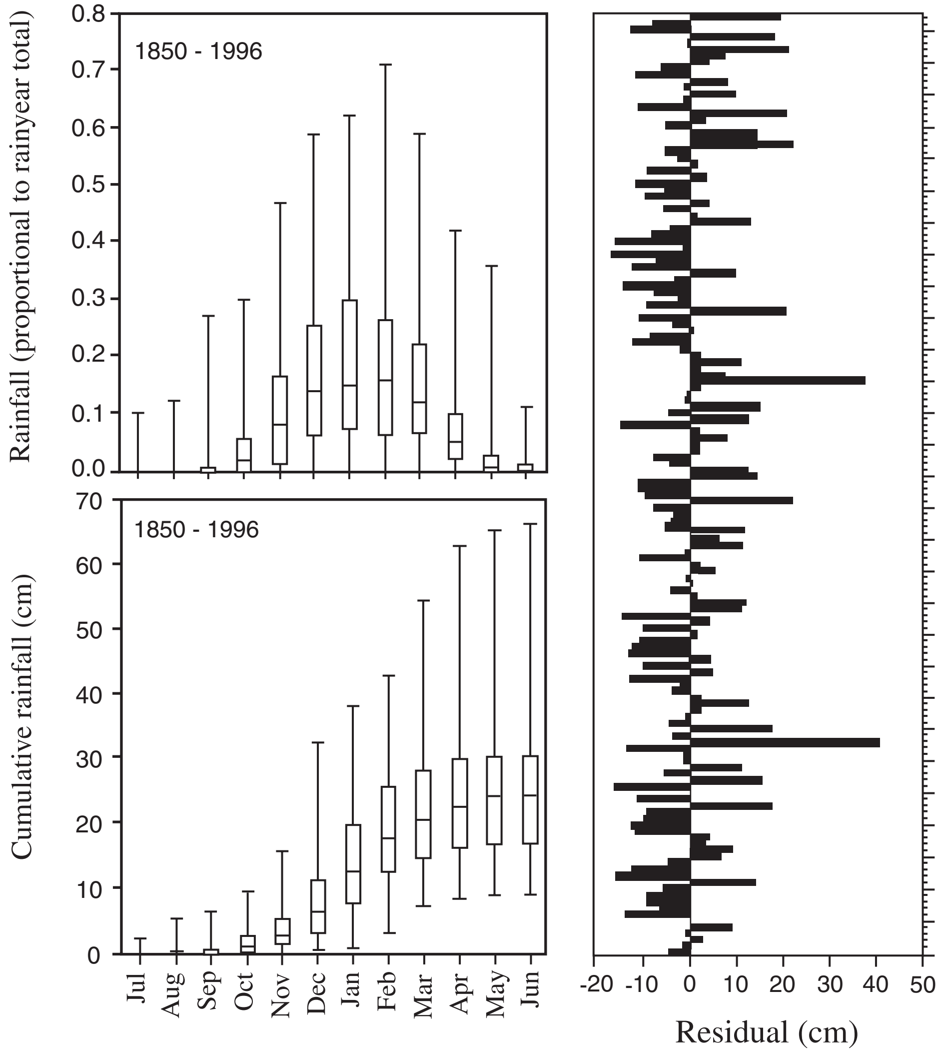

Figure 2.10 Annual and seasonal variation in rainfall for San Diego County. Boxplots include the mean, quartile, and range. (Noe 1999. Abiotic effects on the annual plant assemblage of southern California upper intertidal marsh: does complexity matter? Doctoral thesis. San Diego State University, San Diego, California, USA.)

2.4.2 Constraints posed by heterogeneity

While environmental heterogeneity is desirable, creating complex topography and allowing variation to develop might constrain the ability both to define a precise restoration model and to achieve and maintain specific restoration objectives (Mitsch and Wilson 1996, Simenstad and Thom 1996). Many attributes of coastal wetland variability are not under human control. In southern California, rainfall varies annually and seasonally (Figure 2.10), as do several response variables (e.g., soil salinity, plant productivity, nutrient fluxes, tidal influence). Setting precise targets for plant cover and biomass is thus somewhat subjective, as reference data often provide only a "snapshot" specific to a particular time and location (Zedler 1996*a*, Chapman and Underwood 1997). Reference data are likely to be more reliable when collected over longer time periods (>10 yr) and from several

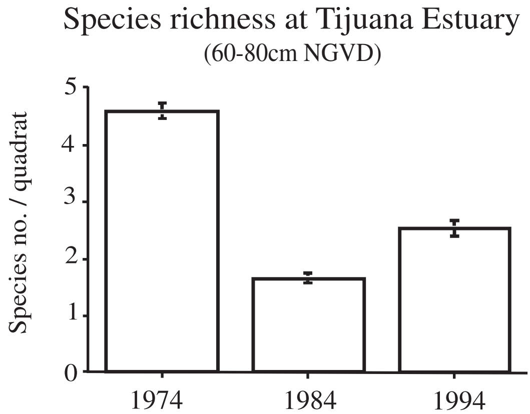

Figure 2.11 Temporal variation in species richness sampled for the marsh plain at Tijuana Estuary. Data are number of species per 0.25 m^2.

reference locations, enabling a more accurate value range for standards to be set. The development of the restoration site can be expected to vary, and the long-term outcome may not be predictable, particularly when monitoring is restricted to the short term (Mitsch and Wilson 1996, Simenstad and Thom 1996, Zedler and Callaway 1999).

Restoration planners need to consider the environmental variation that is likely to occur at the site. One example is the loss of biodiversity associated with development of hypersalinity in coastal remnants or lagoons that experience reduced tidal flows or tidal closure (Ibarra-Obando and Poumian-Tapia 1991, Zedler et al. 1992, Zedler, 1996*a*; Figure 2.11). Tidal closure also leads to water stagnation, harmful algal blooms (Fong 1986, Rudnicki 1986), and low oxygen concentrations in the water, negatively affecting aquatic organisms (fish and invertebrates). In addition, long-term flooding of the salt marsh can reduce habitat available to wading and breeding birds and increase plant mortality due to root-zone anoxia. Drought and hypersaline soils cause mortality of the less tolerant plants and animals (Zedler et al. 1992). The assessment of risks associated with the project must consider the possibility that a wet year might delay earthworks and planting or that a dry year might require irrigation of new plantings. Restoration plans should thus anticipate seasonal and annual variation in environmental conditions and develop contingency plans in the event of untimely storms, floods, or droughts. Plantings can be timed to follow heavy rainfall or precede forecasts of suitable rainfall. The reestablishment of dune vegetation from seed is not worth attempting without some assurance that rainfall will persist long enough to allow seedling roots to tap groundwater. Both seasonal and interannual variations should be considered in the restoration planning process.

The adaptive management approach (Box 6.1) can deal with some of the constraints imposed by environmental variation and unexpected outcomes. It allows for midcourse corrections and adjustments to the plan, so that the risk of failure can be reduced. In adaptive management, adjustments are based on knowledge of how the restoration site is developing. One example is the mitigation program for endangered bird habitat at San Diego Bay (Boxes 1.7 and 1.8). Another is the restoration of tidal flushing regimes through removal of tide gates, levees, dikes, or other restricting structures (e.g., Roman et al. 1984, Simenstad and Thom 1996, Burdick et al. 1997, Streever and Genders 1997, Weinstein et al. 1997).

An understanding of gap dynamics can help restorationists overcome some of the constraints on restoring diversity to salt marsh communities (Figure 2.12). Planting or seeding efforts can be timed to coincide with temporary opportunities, such as periods of

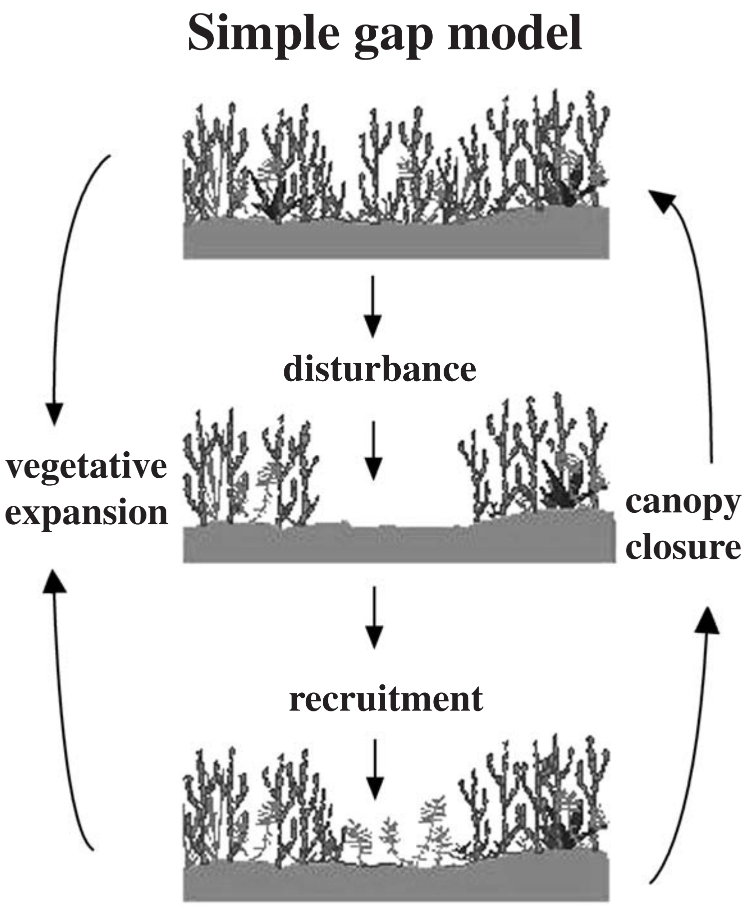

Figure 2.12 Diagram of canopy gap formation, early colonization by a gap-favoring species, such as *Cordylanthus maritimus*, and eventual closure.

lowered salinity. After rainfall or freshwater flooding, soil salinity is lowered, and germination and establishment of seedlings is favored (Zedler et al. 1992, Zedler and Beare 1986, Beare and Zedler 1987, Kuhn and Zedler 1997, Callaway and Zedler 1998). In the high marsh, about 3 cm of same-day rainfall is needed to leach salts from the soil and keep salinities low long enough to stimulate seed germination and allow seedling establishment (Noe 1999) (Figure 2.13; Box 2.2).

Planting or seeding can also be targeted for canopy gaps that are caused by plant mortality, the digging of small mammals (e.g., Cox and Zedler 1986) or tidal deposition of debris that covers vegetation (e.g., Hartman et al. 1983, Bertness and Ellison 1987, Valiela and Rietsma 1995). These gaps may provide the appropriate combination of conditions for germination and seedling establishment (e.g., increased light and soil temperature). This has been tested for the endangered plant, salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*). In the high marsh, maximum diversity of annual species has been correlated with intermediate levels of perennial cover (G. Noe, *unpublished data*). In other studies, recolonization of gaps has been via expansion of primarily clonal species, such as *Spartina alterniflora* (e.g., Hartman 1988). Restoration models should include disturbance regimes that create appropriate canopy gaps.

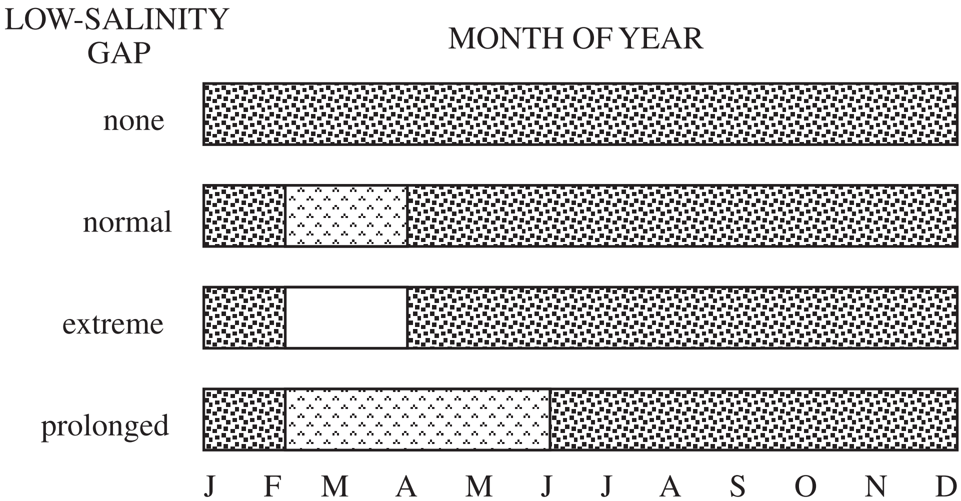

Figure 2.13 Conceptual model of low-salinity gaps in the salt marsh soil. Salinity is indicated by the density of the dots; typical conditions are hypersaline (>40 ppt) (from Zedler et al. 1992, *The ecology of Tijuana Estuary, California: a National Estuarine Research Reserve,* NOAA Office of Coastal Resource Management, Sanctuaries and Reserves Division, Washington DC.). Opportunities for successful plant germination and establishment are likely to be higher during times when the salinity gap exists.

2.5 *Constraints posed by human use*

2.5.1 *Human concerns*

The historical use of a site by Native Americans may constrain its potential for restoration. If cultural artifacts are present, the evidence of earlier uses could become a cause for celebration, as well as a point of contention. Involvement of groups and individuals with knowledge of early cultures can help identify which sites should not be disturbed and pinpoint the opportunities for interpretation once the restoration is underway. At Tijuana Estuary, a film about the wetland and its need for restoration includes Native American speakers who describe the significance of the coastal wetland to their ancestors.

Because tidal wetland restoration sites are often in or near urban areas, the concerns of local residents must be taken into account in setting targets. Failure to consider the opinions of an admiring or critical audience can have serious consequences. In the now-famous Chicago example (Stevens 1995), the efforts of avid savanna restorationists to reduce tree densities through cutting and burning were thwarted for several years while they argued with other environmentalists about what was and was not desirable. The critical instrument was the permit for controlled burning, and decisions to grant permits were made in part on the basis of citizen input. In the face of loud opposition, few decision-makers will risk sharp criticism; rather, they will delay and deliberate, and restoration will be put on hold. The strong lesson from this experience is that the public needs to be consulted, informed of, and drawn into the restoration process.

An effective way to involve local residents is to identify members of existing nongovernmental organizations who have concerns about either the site in question or its biota.

There is usually a strong network of individuals belonging to the various conservation organizations, and it takes little time to identify key people who can come to meetings and serve as information disseminators. Organizational newsletters are also effective means of communication. In southern California, virtually every coastal wetland has a "Friends of …" organization, and the early involvement of members helps enormously in planning and implementing restoration efforts.

2.5.2 *The desires of local residents*

People care about their surroundings, but opinions on what they should look like are as numerous as the individuals who express them. Hence, restoration plans will need to be presented clearly, and the appearance of the restoration model sketched (e.g., concept plans and drawings) with attractive visual aids. Project proponents can develop graphics that summarize the problems that restoration will solve and how the site will change and improve. They can then call for suggestions from the public and suggest how suitable ideas can be built into the project. If items of contention are uncovered, it will be wise to identify them early and seek to understand their roots, as well as to work with opponents to develop mutual understanding of the opportunities afforded by restoration.

At Tijuana Estuary's Tidal Linkage (Box 1.10), a major concern was whether or not the newly excavated tidal channel should have a bridge to allow visitor access to tidal ponds where the public was accustomed to viewing shorebirds. Some environmentalists preferred to use the channel as a barrier to uncontrolled human access. Others wanted to enhance visitor amenities by making the bridge into an interpretation opportunity, as the site is near the Visitor Center and trail origin. Federal and state managers of the site and visitor center preferred the bridge option, which was presented as a compromise — the new channel would cut off most of the ingress, focusing it at one point (the bridge), where human activities could be regulated. Access was viewed as a major benefit to the public, whose support was needed for this extremely costly restoration project to go forward with public funding.

In most urban settings, maximum protection through exclusion of the public is not an option. Compromises are the rule, with some access provided in exchange for other conservation measures. The task is thus to find new and innovative kinds of access while minimizing negative consequences. Viewing blinds, telescopes, creative display panels, and guided activities may satisfy curiosity while confining people to the edge of the wetland and, at the same time, increasing awareness of how human activities may negatively impact coastal wetlands. The Tidal Linkage (Box 1.10) owes its existence to the idea that the marsh should be brought to the people, rather than bringing the people into the marsh. That is, instead of building a boardwalk into the natural salt marsh, where endangered birds were known to nest and forage, a project was conceived to provide an even better viewer experience, namely a complex of open-water and marsh habitats. The channel and marsh and their biota are now on full view to visitors, without impacts to natural nesting habitat.

2.5.3 *The constraints posed by an adjacent human population*

An adoring public can be a major plus for a restoration project, but people need to be educated about the fragile nature of salt marsh vegetation and the sensitivity of marsh animals. Clearly marked trails, signs about sensitive species, and rope fences may be necessary to protect restoration sites, especially in the early, bare stages, when values are hard to appreciate. One option is to create labeled displays of principal plant species near

the trail or in pots near a visitor center. Large areas of plants with showy flowers might not be natural, but they might improve the visitor experience and change public attitudes. A second option is to create an effective buffer between the restoration site and visitor access points. In southern California, a native shrub, *Lycium californicum*, and a spiny rush, *Juncus acutus,* both offer possibilities for reducing visitor access (Box 2.3). The shrub is extremely thorny and the rush has sharp leaves. To avoid liability in case of accidental injury, it would be wise to plant non-thorny vegetation next to the trail, with signage indicating that thorny plants occur just off-trail.

Vandalism is another constraint on urban wetlands, and no reasonable level of policing can prevent it. The options for reducing vandalism are education, involving local people of all ages in the project, creating a sense of community pride in the project, and training volunteer rangers to speak to offenders and perhaps even enlist their help. Education programs developed for local schools can change attitudes about places once considered wastelands.

2.6 Stepwise planning of restoration

Ideally, background information gathered on restoration needs and priorities within the region should form the backbone of a restoration model. Once goals have been clearly identified, potential restoration sites can be assessed for their suitability in meeting these goals, with restoration constraints and opportunities posed by the sites identified. Work can then begin on a specific restoration plan that seeks to meet these goals (Zedler 1996*a*,*b*). This will involve developing conceptual models (e.g., layout and interrelationships of habitat types), specific protocols (e.g., soil amendment protocols, grading specifications), target parameters (percent vegetative cover, animal usage), and management and monitoring programs.

The development of a restoration model is a stepwise, iterative process whereby multiple alternatives can be considered and evaluated. Ideally the restoration framework should aim to create self-sustaining habitats that can deal with changing environmental conditions. Adaptive management will allow goals, assessment, and management procedures to be reassessed depending on the performance outcomes of the project. The restoration plan should also be amenable to scientific monitoring and assessment of the goals or targets. Experimentation to test restoration protocols and our understanding of wetland functioning should be done in addition to basic monitoring and assessment (Box 2.4). This both tests and builds upon our current knowledge of the ecosystem reassembly process (see PERL 1990).

Developing the restoration model should take into account both regional and local restoration needs (Figure 2.14) (Zedler 1996*b*). While restoration targets will vary considerably from one region to another, a regional analysis of needs and targets involves the evaluation of historical wetland losses (including habitat, species and functional losses) and an assessment of the regional resource base through detailed inventory of the remaining wetland habitats (e.g., habitat type, location area and condition, and presence of significant species). These initial steps allow for the establishment of targets (e.g., based on habitat representation, condition, and prevalence of endangered species) with subsequent identification of opportunities to do restoration and enhancement (e.g., expansion of habitat, expansion of targeted native species populations, maintenance or improvement of biodiversity or function). Following from this, the specific restoration and mitigation requirements should be matched with the various sites' opportunities and regional targets. It is this matching of regional targets with local opportunities and requirements that shapes the setting of local (site-specific) restoration goals.

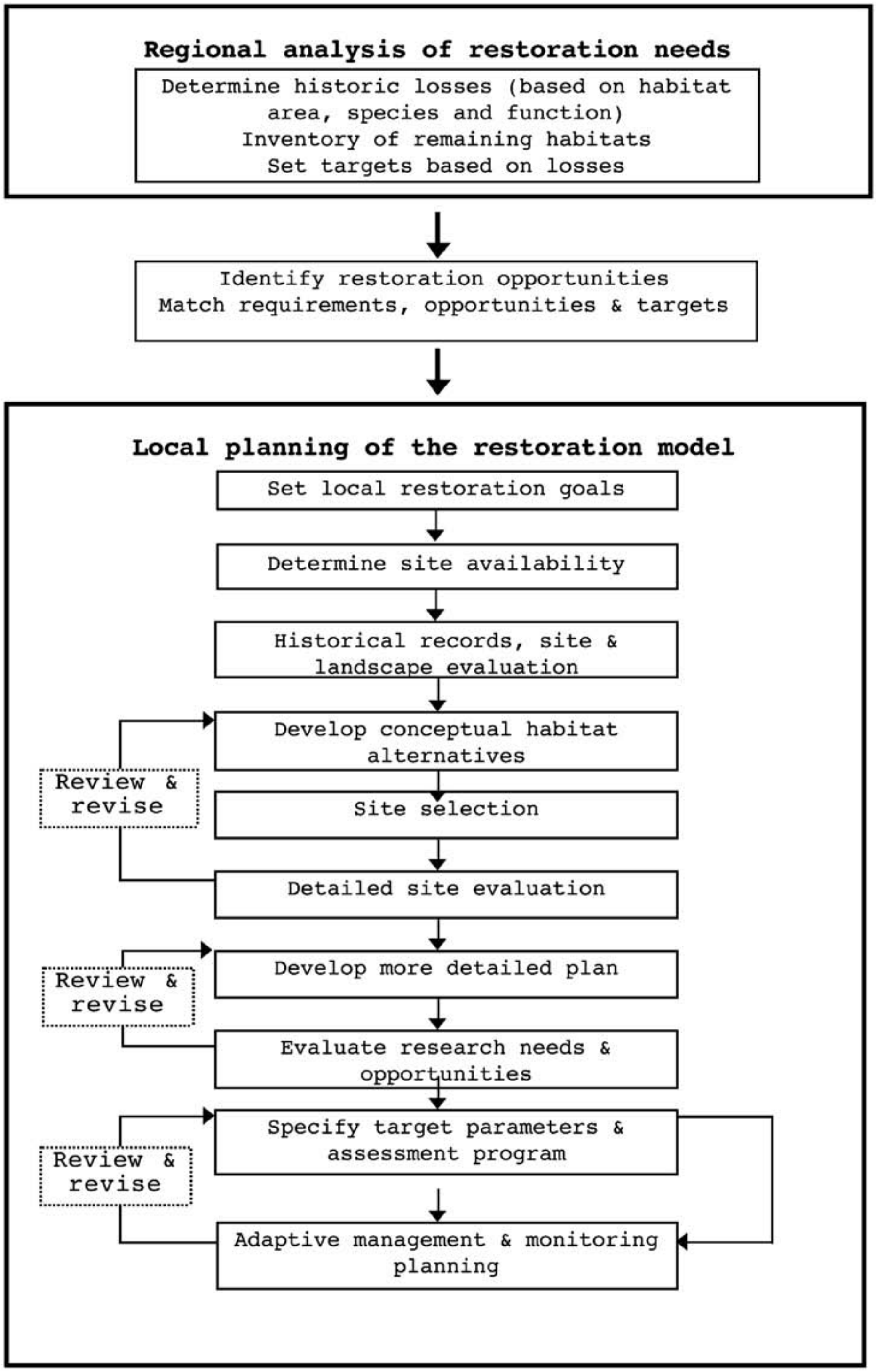

Figure 2.14 Flow chart outlining the various steps involved in developing a detailed restoration plan.

Local analysis is a more complex planning process involving the steps outlined in Figure 2.14. It is important that the restoration goals be clearly defined at the beginning of the process, as these will mold the later steps. Identification of landscape issues associated with potential sites should focus on heterogeneity, area, connectivity, configuration, and spatially interactive processes with adjacent ecosystems (upland, riparian, and marine). The development of conceptual habitat alternatives includes habitat choice, area, and layout. At some sites, several alternatives may be possible (e.g., different proportions of marsh plain, mudflat, and high marsh) while at others the existing elevations and costs of grading and removing fill may restrict the habitats to a single habitat type or a fixed combination of habitats (e.g., 20% marsh plain, 70% high marsh, 10% upland:wetland transition). Site selection ideally balances the opportunities and constraints of the potential sites. Detailed site evaluation will assess hydrological, soil, and other biotic conditions (collection of preliminary data: soil cores, species lists, etc.). Review and revision of earlier concept design and site selection follows detailed site evaluation. This is particularly important when site evaluation identifies potential constraints such as high contaminant loads in soils/sediments or the presence of endangered species. Once the concept plans are firm, it is advisable to seek feedback from the local community by putting them on display at the local environmental center. Such consultation with the local community at this stage is a useful way to gain further input (e.g., local knowledge of the site), increase a sense of community ownership and participation in the project (e.g., they may later act as volunteers to help in maintenance), and anticipate or head-off negative reaction.

Developing a more detailed site plan also involves drafting financial budgets, grading plans, and planting plans identifying the most appropriate restoration procedures. This will involve selection of revegetation methods (e.g., identifying plant sources, seeding, replanting, encouraging natural colonization) and animal recolonization strategies (e.g., natural recolonization or reintroduction). Detailed plans should not be developed until the conceptual plan and location are firm, as plans are costly to change. Many specialists will need to be involved in plan development and implementation, including environmental engineers, hydrologists, ecologists, town planners, surveyors, land managers, and landscape architects. While consulting with these individuals, it is useful to evaluate research opportunities and to design experiments into the plan at an early stage, particularly if large areas can be devoted to testing different restoration procedures.

The definition of restoration targets (ranges of acceptable outcomes) and assessment protocols (what, when, and how the site will be monitored) will largely depend on the local restoration goals. An adaptive management approach should (1) build on experiences at other sites; (2) review progress at the site; and (3) amend goals or management procedures where necessary.

Box 2.1 Reference data for use in restoring Tijuana Estuary

Gabrielle Vivian-Smith

A study of a near-pristine coastal wetland (Volcano Marsh on San Quintín Bay, Box 1.12) in Baja California Norte, Mexico, provided both reference data for a restoration framework and evidence that topographic heterogeneity structures the salt marsh vegetation (Zedler

et al. 1999). The central question of the study asked whether elevation is an adequate descriptor for the environmental conditions determining plant species distributions. Plant species occurrences were related to elevation, proximity to the bay, and proximity to tidal creeks by recording species occurrences, elevation, presence of tidal creeks or channels <1 m from the quadrat, and proximity to the bay, in over 800 points along randomly located transects oriented from the bay through the high marsh.

Findings

Elevation did not adequately describe plant species occurrences. The traditional zonal models of salt marsh vegetation, with tight species associations correlated to an elevation range, were very generally applicable (e.g., Nixon 1982, Vince and Snow 1984, Pennings and Callaway 1992, Sanchez et al. 1996). While many conceptual drawings of salt marshes show a gradually sloping topography, the surface at Volcano Marsh was largely flat, but topographically complex due to the many creeks dissecting the marsh surface (Figure 2.15). Topographic complexity (measured by the numbers of tidal creeks) also differed between wetland areas above and on the marsh plain. Comparisons with elevation surveys at Sweetwater Marsh in San Diego Bay also show large areas of relatively flat topography (Figure 2.12B).

Plant species occurred in broadly overlapping bands with distributions further complicated by smaller-scale heterogeneity (microtopography and tidal creeks). The tidal creek margins tended to have more species, due to the fact that several species (*Salicornia bigelovii, Suaeda esteroa*, *Limonium californicum* and *Frankenia salina*) occurred at lower elevations when growing along these margins (Figure 2.16). In many of the transects, one species, *Spartina foliosa*, was found closer to the bay despite suitable elevations further inland (Figure 2.15, see Transects b, c, and e). Three general habitat types could be characterized by combining both elevation and the occurrence of conspicuous plant species: (1) high marsh, a 30–70 cm elevation range with *Salicornia subterminalis*; (2) the marsh plain, a 30-cm elevation range with heterogeneous topography and nine common species; and (3) cordgrass habitat, the bayward portion of the marsh plain and lower elevations with *Spartina foliosa*. Most species were found on, but were not restricted to, the marsh plain.

Recommended applications

1. Restore habitat types in their appropriate spatial locations. High marsh can be graded slopes of various angles between upland areas and the marsh plain. Marsh plain can be developed by creating a broad flat plain with cross-cutting creeks for topographic complexity. Areas for cordgrass can be developed by providing channel edge habitat and by planting *Spartina foliosa* near open water at suitable elevations.
2. Restore or create tidal creek networks to provide topographic complexity. Physical characteristics of constructed or restored coastal wetlands should include tidal creeks, at similar configurations and densities to those in natural coastal marshes. The resulting habitat complexity should enhance species richness and provide habitat for other organisms reliant on tidal creeks, such as juvenile killifish (Desmond 1996) and crabs, a food source for the light-footed clapper rail (Jorgensen 1975). Such creek or channel systems can be modeled on natural systems (e.g., by studying drainage patterns of reference sites from aerial photographs and visual inspection) and be incorporated into the grading plans for the site.

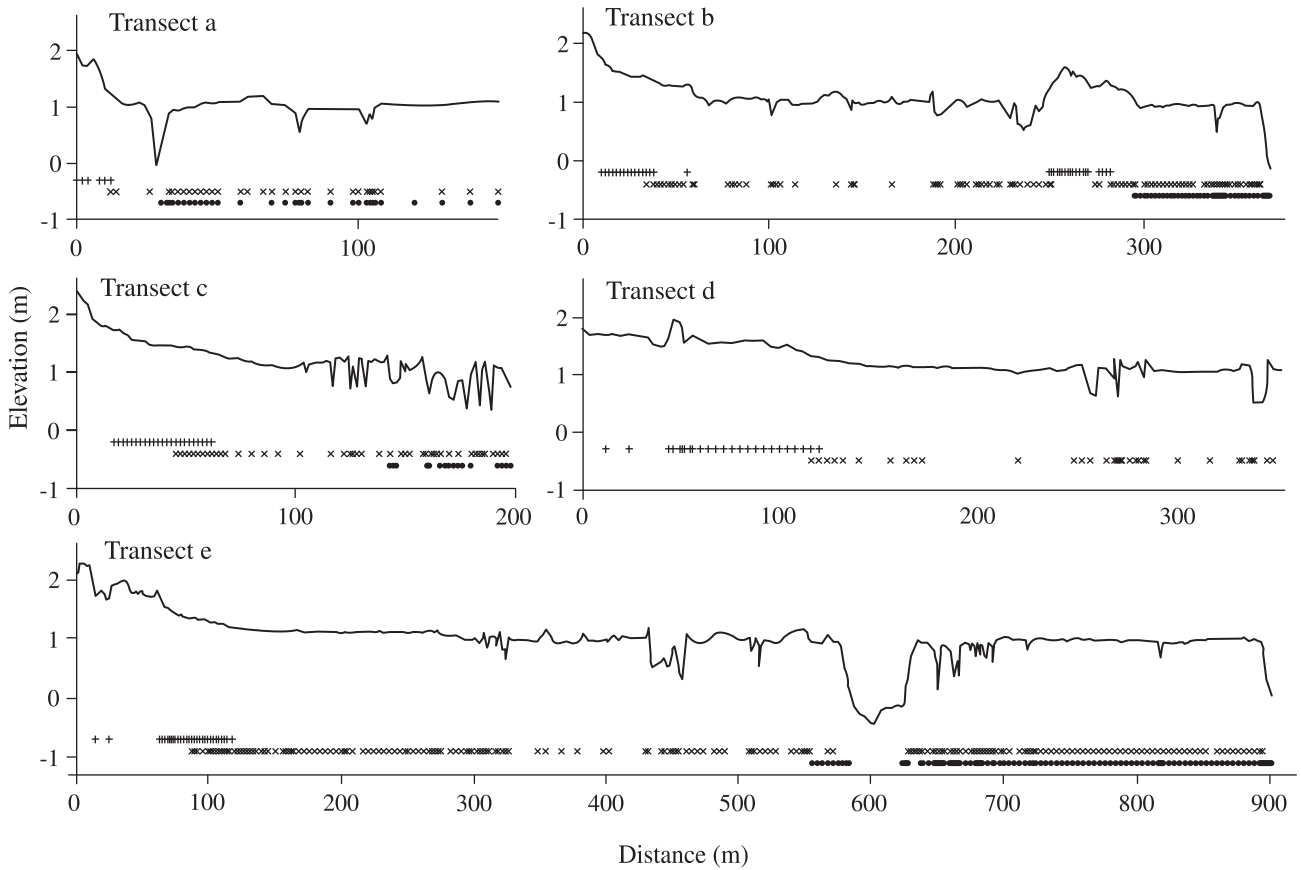

Figure 2.15 Transect elevation profiles at Volcano Marsh, San Quintín Bay, Baja California, Mexico, showing occurrences of *Salicornia subterminalis* (+), *Salicornia virginica* (x), and *Spartina foliosa* (•). Elevations are relative to the lowest occurrence of *Spartina foliosa* (from Zedler et al. 1999, Californian salt-marsh vegetation: an improved model of spatial pattern. *Ecosystems* **2**:19-35, with permission).

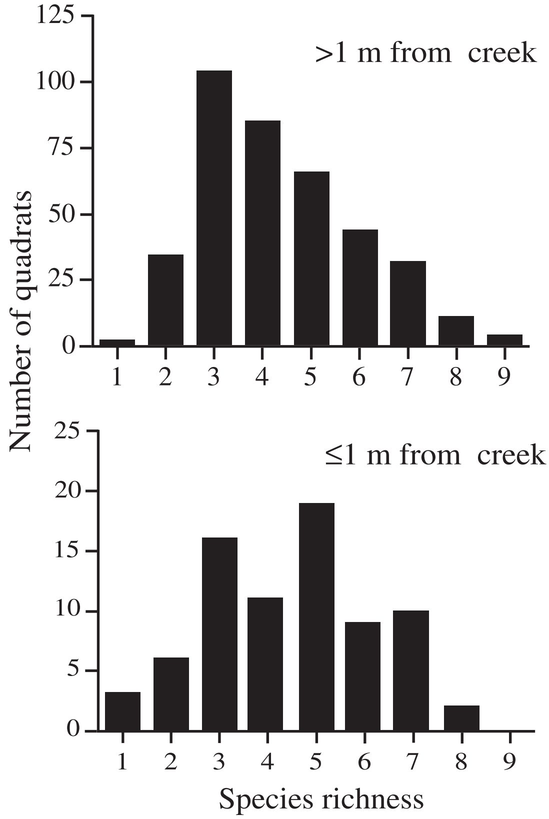

Figure 2.16 Frequency distribution of species richness near (less than 1 m) and away from (1 m or more) tidal creeks on the marsh plain at San Quintín Bay, Baja California, Mexico (from Zedler et al. 1999, Californian salt-marsh vegetation: an improved model of spatial pattern. *Ecosystems* **2**:19-35, with permission).

3. Plan elevations for planting based on those found in natural marshes. Species elevation ranges in natural marshes are broader than those likely for plants growing in a newly constructed habitat, particularly those lacking tidal creek networks. To maximize survival, establishment, and subsequent colonization of plants at the restoration site, species should be planted closer to the modal elevations they occur at and not the extremes (Figure 2.17). *Spartina foliosa* plantings should be located at suitable elevations and in closer proximity to the bay or tidal inlet than other species.

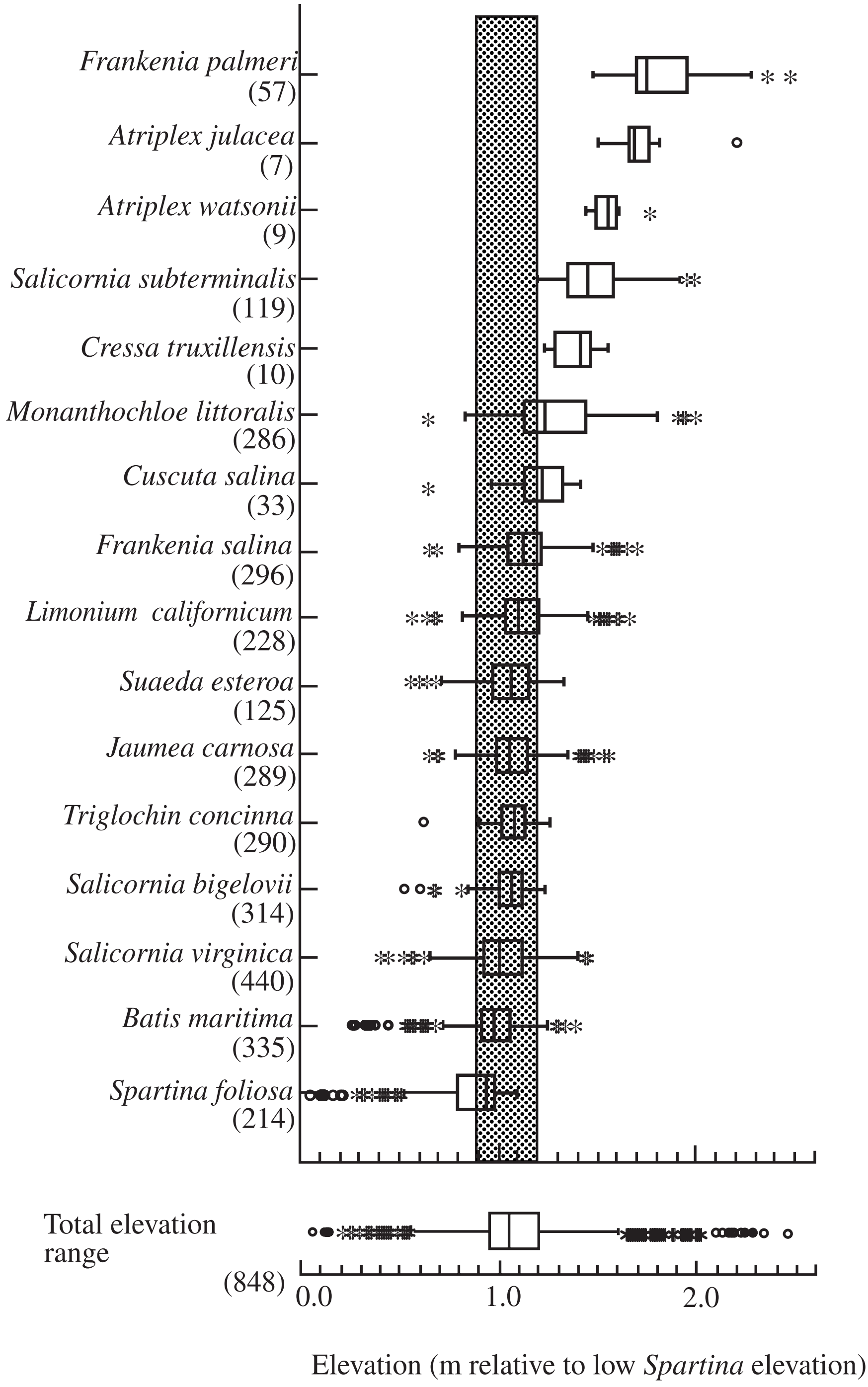

Figure 2.17 Elevation range boxplots of individual species and the total sample at Volcano Marsh, San Quintín Bay, Baja California, Mexico (from Zedler et al. 1999, Californian salt-marsh vegetation: an improved model of spatial pattern. *Ecosystems* **2**:19-35, with permission). The boxes and attached bars (or whiskers) indicate the elevation range where most individuals of each species were found, with the central bar representing the modal elevation. The number of encounters is indicated in parentheses.

Box 2.2 The importance of small-scale heterogeneity to high marsh annual plants

Gregory B. Noe

Patterns

Small-scale spatial variation structures the high salt marsh plant community in southern California, with seedlings of annual species responding to differences in soil moisture, soil salinity, texture, elevation, and perennial canopy cover (Noe 1999). In the field, conditions vary at several spatial and temporal scales, creating patchy distributions of vascular plants, especially the annual species, which have small root masses that cannot integrate conditions over large areas. In the brief wet season (rain falls sporadically between November and March in southern Calfornia), salinity and moisture change with major rainfall events. While all the annuals germinate during the cool wet season, the species germinate at different times and places in response to various combinations of salinity and moisture (Figure 2.18). Both the temporal and spatial variability in salinity and moisture are important to seedling establishment.

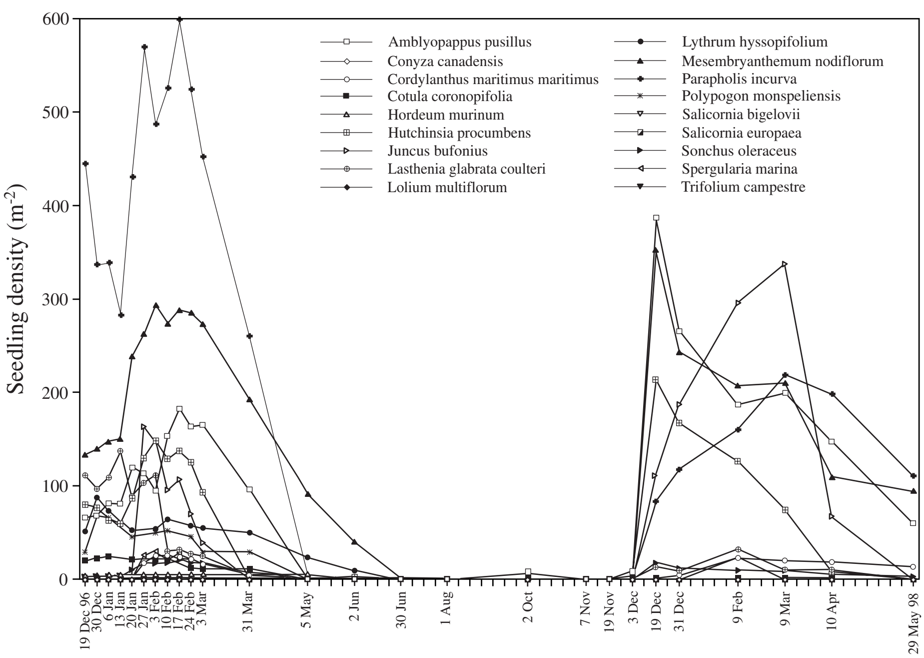

Figure 2.18 At Sweetwater Marsh, Tijuana Estuary, and Los Peñasquitos Lagoon, annual species are separated by soil moisture and salinity gradients (Noe 1999).

Exotic and native annuals

Exotic species contribute most of the individual annual plants found in southern California salt marshes. At Tijuana Estuary, over 90% of the seedlings found in 1997 were exotics, primarily the annual grass, *Parapholis incurva*. The exotic species tend to occur in areas with high soil moisture and low soil salinity, while the native species are mostly found in areas of low soil moisture and over a broad range of salinities (Zedler and Beare 1986, Noe 1999). Thus, freshwater runoff into a restored marsh and prolonged irrigation can be expected to encourage exotic and weedy species (Zedler et al. 1990*a,b*, Kuhn and Zedler 1997, Callaway and Zedler 1998, Noe 1999).

Box 2.3 Ecotones in coastal wetland restoration

Matt James

Importance of ecotones

Ecotones are narrow transition zones or interfaces between more widely distributed habitat types. Ecotones have significance as buffers between terrestrial and aquatic systems (Holland et al. 1991), influencing both the functioning and diversity of coastal wetland systems (Neuenschwander et al. 1979, James and Zedler 2000). The wetland-upland transition zone provides a unique habitat that is differentiated from coastal shrub communities due to the influence of salt. Due to losses associated with urban development (infilling and construction), only a small and often disturbed fraction of the historical extent of this habitat remains. Despite extensive losses, revegetation of this habitat is rarely given consideration in restoration. In southern California, this ecotone is important to sensitive bird, mammal, and reptile species (Zedler et al. 1992). This zone can also regulate both sediment and nutrient flows from the upland to the wetland (Holland et al. 1991, Johnston 1993, Risser 1993).

Applications to restoration

When creating an upland-wetland ecotone, both the slope and planting specifications are important. Gentle slopes, similar to those typically found in natural transition zones, should be planned. This will both maximize habitat value and encourage natural wetland functioning. Most coastal restoration efforts in southern California do not follow this natural model. Instead, the edges of constructed (often excavated) wetlands have steep grades from high marsh to upland, usually all within the same slope angle. Slopes of 33% were constructed at the Tidal Linkage and are planned for the Model Marsh at Tijuana Estuary (Box 1.11). These maximize the area of wetland within a project area; at the same time, they minimize the area available to restore populations of ecotone species. Steep transition zones also erode during heavy rainfall and high storm tides, allowing the adjacent marsh to fill in with sediment (Zedler 1996*a*). Gradual slopes enhance ecosystem function by trapping nutrients from the surrounding upland and by providing more extensive habitat for ecotone species, such as the thorny subshrub, *Lycium californicum* (James and Zedler 2000). At Tijuana Estuary, the natural transition zone slopes range from <1% to >7%. Restored wetlands should have at least some areas with gradual slopes.

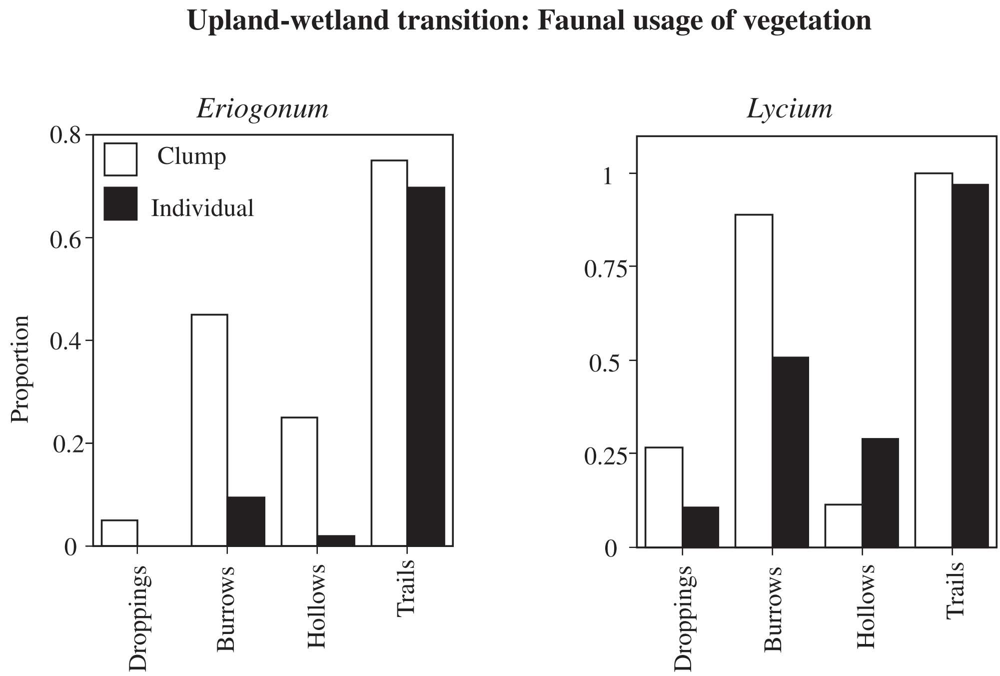

Figure 2.19 Vegetation structure and evidence of faunal usage in the upland:wetland transition at Tijuana Estuary (James 1998, *Dynamics of wetland and upland subshrubs at the salt marsh-coastal sage scrub ecotone,* Master's thesis, San Diego State University, San Diego, CA.). Data are proportion of sample points where evidence was seen.

Habitat values for animal species can be enhanced by selecting and planting ecotone species that emulate natural patch structure. In southern California, *Lycium californicum* is an especially important species for animals because it provides dense shrubby cover and vertical structure immediately adjacent to the high marsh. Birds use the canopy for nesting, for perching, and as a high tide refuge near the marsh (Figure 2.4). *L. californicum* and *Eriogonum californicum* should be planted both in monospecific clumps (<5 m diameter) to maximize small mammal habitat (Figure 2.19) and also as scattered individuals to provide cover for animals moving between habitat patches (James 1998).

Box 2.4 Evaluating disturbance effects to aid wetland restoration

John Callaway

The traditional method of evaluating restored wetlands is to compare ecosystem structure (and occasionally function) with that of natural wetlands. We also promote the evaluation of ecosystem responses to disturbance events, whether planned as an experiment or unplanned.

Experimentation with planned disturbance

At Tijuana Estuary, we constructed tidal mesocosms to test the resilience of planted salt marsh vegetation to hydrologic modifications (Callaway et al. 1997). Exotic plant invasions occurred readily where soil salinities were lowered by freshwater treatments. Only the mesocosms with saline soils were resistant to invasion. These findings relate directly to the issue of restoration progress; i.e., sites with excess freshwater inflow can expect greater problems with invasive exotics.

Responses to unplanned disturbance

The second approach, evaluating ecosystem responses to unplanned disturbance events, can provide valuable information for management of restoration sites. Documenting such responses to disturbances can provide a greater understanding of ecosystem and individual species responses to changing environmental conditions. For example, an algal bloom at the Tidal Linkage (Box 1.10) allowed assessment of the resilience of different macrophyte species and assemblages to smothering by *Enteromorpha* and *Ulva* species. Other typical unplanned disturbances at restoration sites include herbivory, sedimentation and erosion events, exotic species invasions, and drought. Because the ideal for restoration projects is to create sustainable ecosystems over the long term, observing ecosystem responses to either planned or unplanned perturbations can provide insights into the ecosystem's resilience and recovery mechanisms. Such responses can help us expand our understanding of coastal wetland restoration by prompting us to formulate questions or hypotheses that can then be evaluated using more rigorous replicated experiments.

Comparing the disturbance responses of restored, engineered, and natural wetlands may yield valuable information about the development of resilience, resistance, and recovery mechanisms at these sites. (Engineered wetlands have structures regulating their tidal flows or freshwater inflows.) Damages due to natural disturbances, and a mitigator's liability for such damages, can best be determined by simultaneously assessing the degree of change in constructed and natural wetlands. A mitigator might not need to be required to construct a wetland that is free of disturbance; rather, the engineered system should be requred to be at least as resilient to the disturbance as a reference wetland. Optimally, tests for engineered and natural wetlands should uncover shortcomings in functioning (e.g., experiments with nutrient loading to determine growth-limiting factors) and observations of wetland responses to invasive species and flooding (e.g., impacts of erosion and sedimentation events).

Box 2.5 The diversity of habitats in southern California coastal wetlands

Julie S. Desmond, Gregory D. Williams, and Gabrielle Vivian-Smith
Illustrated by Donovan McIntire

Subtidal habitat

Estuarine habitats with subtidal configurations include harbors and sheltered bays with open water. Subtidal habitats are inundated throughout the tidal cycle and have less

variable physical environments than intertidal areas. Furthermore, they are generally located in close proximity to the ocean, where environmental conditions (e.g., salinity and temperature) often reflect an intermediate level between deep water ocean and shallow intertidal habitats. Seagrass beds are a marine wetland habitat that is frequently found in shallow subtidal areas fringing coastal wetlands. They support fish species that inhabit tidal creeks and channels and salt marshes. A common submerged plant species in southern California is eelgrass (*Zostera marina*). In southern California, subtidal basins have replaced shallow, marsh-dominated estuaries as the main inshore habitat for fish because of extensive dredging and filling (Horn and Allen 1985).

Intertidal flats

Intertidal mud- and sandflats are devoid of vascular plants but can have significant algal biomass. Mudflats provide habitat for the most diverse invertebrate assemblages of the coastal wetland; these assemblages include gastropods and bivalves, polychaetes, amphipods, and crabs (Levin et al. 1998; Figure 2.20). Flats also act as valuable feeding and resting sites for many species of shorebirds. Boland (1981) found that intertidal mudflats were used by many more shorebirds (both species and individuals) than any other habitat type at Tijuana Estuary. The different sediment depths and water depths are thought to provide a wide variety of prey items supporting many species of birds.

Tidal creeks and channels

Complex networks of large channels and small creeks carry tidal flows between emergent vegetated marshes and subtidal habitats. Tidal channels and creeks provide a variety of habitats varying in morphology (Chapter 5) and other characteristics, including water depth and salinity, sediment textures, water movement, temperature, nutrients, and dissolved gases. Creeks and channels support many bird, fish, and invertebrate species as well as macroalgae and phytoplankton (Zedler et al. 1992). The major factors involved in their formation are the tides, the water volume (or tidal prism), slope, sediment type and consolidation, and vegetation (Eisma et al. 1997). Below, we contrast the more narrow first- and second-order intertidal creeks ("creeks") with the wider third- and higher-order subtidal channels ("channels").

First- and second-order creeks average approximately 1 and 4 m in width, respectively (Odum 1984), and they usually drain completely at low tide. In comparison to large, high-volume channels, these creeks have lower tidal flows and higher evaporative rates. This influences their physicochemical conditions; i.e., salinity, temperature, and dissolved oxygen concentrations fluctuate more in first-order creeks than in larger channels (Desmond 1996). They also are occasionally vegetated, with high relative edge:volume ratios (Figure 2.21). First- and second-order creeks generally comprise a majority (65 to 85%) of total channel length in southern California marshes (Pestrong 1965, Coats et al. 1995, Desmond 1996).

Channels average approximately 10 m in width (Odum 1984), carry relatively high flow volumes, and are generally distributed closest to marine habitats. Because they remain inundated during all tidal stages, they can support submerged aquatic vegetation, such as eelgrass (Figure 2.22). Up to 15 to 35% of the total channel length in many California marshes is third- and fourth-order channels (Pestrong 1965, Coats et al. 1995, Desmond 1996).

Figure 2.20 Mudflat habitat with shorebirds in the foreground, cordgrass in the background, and terns overhead. McIntire Drawings, © 1999 by Zedler.

Figure 2.21 Intertidal creeks and salt marsh, with the Belding's Savannah sparrow. McIntire Drawings, © 1999 by Zedler.

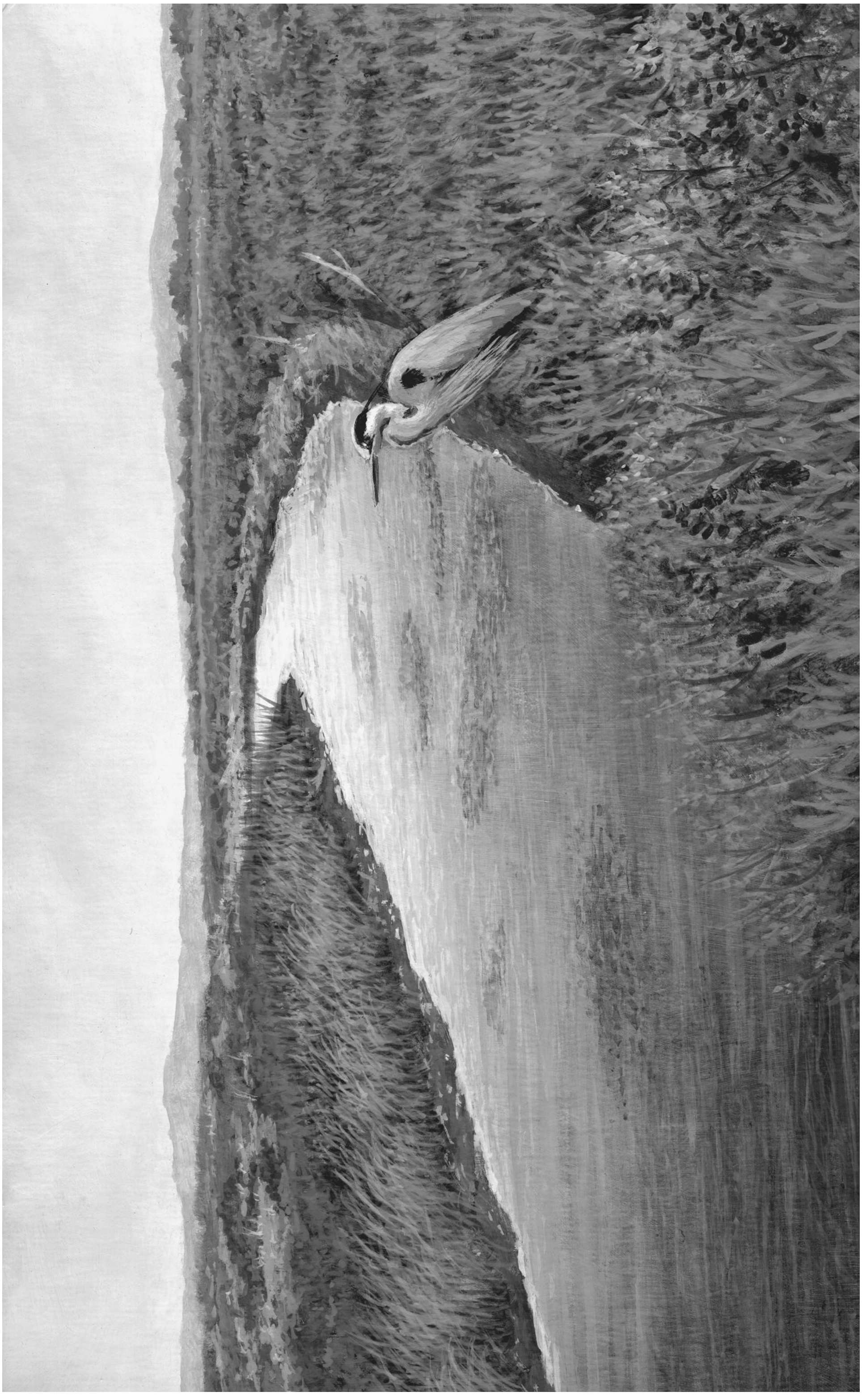

***Figure* 2.22** A large tidal channel with a great blue heron at the salt marsh edge. McIntire Drawings, © 1999 by Zedler.

Salt marsh

Salt marsh is a plant-dominated intertidal habitat supporting a wide variety of invertebrate and vertebrate species. In southern California, salt marsh habitat characteristically has up to three different plant assemblages: cordgrass marsh, marsh plain and high marsh. The range and relative areas of these habitats vary, with cordgrass marsh found more commonly at lower elevations and with full tidal flushing. Salinity levels of salt marsh habitats can vary considerably and are influenced by seasonal rainfall variation, tidal flooding frequency, elevation, soil depth and texture, and canopy cover. Typical salinity levels range from 40 to 100 ppt. Plant species form broadly overlapping bands intersected by tidal creek networks (Figure 2.16). Cordgrass marsh is dominated by cordgrass (*Spartina foliosa*), a largely clonal species that dominates the lower elevations at the bayward portions of the marsh. Cordgrass forms important year-round habitat for an endangered bird species, the light-footed clapper rail (*Rallus longirostris levipes*).

Frequently abundant species of the southern California marsh plain (Figure 2.23) include pickleweed (*Salicornia virginica*), salt wort (*Batis maritima*), annual pickleweed (*Salicornia bigelovii*), sea-blite (*Suaeda esteroa*), fleshy jaumea (*Jaumea carnosa*), arrow-grass (*Triglochin concinna*), and sea lavender (*Limonium californicum*). While some species are widespread (e.g., *Batis maritima*) and frequently abundant (e.g., *Salicornia virginica*), many of the other species occurrences and abundances vary from marsh to marsh (Appendix 3). Pickleweed (*Salicornia virginica*) provides habitat for the endangered Belding's Savannah sparrow.

Characteristic perennial plant species of the high marsh include perennial glasswort (*Salicornia subterminalis*), shoregrass (*Monanthochloe littoralis*), alkali heath (*Frankenia salina*), salt grass (*Distichlis spicata*), sea lavender (*Limonium californicum*), Watson's saltbush (*Atriplex watsonii*), and *Cressa truxillensis*. Many annual plant species exploit the wet and low-salinity winter conditions in the high salt marsh, including at some locations the sensitive species Coulter goldfields (*Lasthenia glabrata coulteri*). During this period two annual plant assemblages develop under sparse perennial canopies and at the salt panne edge. These assemblages occur at similar intertidal elevations. The salt panne edge (approximately 2 m around the panne) also has sparse cover of perennial plants. The salt panne edge has higher soil salinity, lower perennial cover, and shorter canopy height than the sparse perennial canopy habitat, but soil moisture is similar. The endangered salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*) occurs in the high marsh plant community.

Salt panne

Salt pannes are upper intertidal salt flats that lack vegetation (Figure 2.24). The duration of inundation varies from weeks to months depending upon the amount and timing of rainfall. Pannes develop a salt crust during the dry summer, when evaporation typically drives soil salinity levels up to 200 ppt. Algae are not conspicuous at this time, but invertebrates such as rove beetles (*Bledius* spp.) can be found burrowed in the sediments. Some bird species, such as California least terns (*Sterna antilarrum browni*) and snowy plovers (*Charadrius alexandrinus nivosus*), nest on the dry panne surface. During the short, wet winter season salt pannes temporarily become aquatic systems, accumulating rainwater and saline water and providing feeding and resting sites for ducks and snowy plovers (Zedler et al. 1992). Under these conditions algae and ditch grass (*Ruppia maritima*) can often be found in the shallow pools. This temporal variation in conditions provides very different habitat values throughout the year. Salt panne habitats also support threatened

Figure 2.23 Marsh plain habitat with several succulent halophytes. McIntire Drawings, © 1999 by Zedler.

Figure 2.24 Salt panne habitat and patchy high marsh vegetation, with a killdeer on the right and rove beetle middens in the foreground. McIntire Drawings, © 1999 by Zedler.

species of tiger beetles (*Cicindela* spp.) and are often used by foraging Belding's Savannah sparrows (*Passerculus sandwichensis beldingi*).

Brackish marsh

Brackish marshes have salinities lower than seawater (0.5 to 30 ppt) due to their location near freshwater springs and seepages. They tend to develop where urban and agricultural runoff collects, as well as where wastewater spills are impounded. In some salt marsh habitats, the discharge of less saline water allows brackish marsh species to invade and persist, temporarily or permanently displacing salt marsh vegetation (Beare and Zedler 1987). Water levels can fluctuate erratically where freshwater inflows are impounded, and so can salinities, as flooding and evaporation alternate. Plants typically occurring in brackish habitats are cattails (*Typha domingensis*), bulrushes (*Scirpus californicus*), ditchgrass (*Ruppia maritima*) and spiny rush (*Juncus acutus*) (Figure 2.25). Freshwater and brackish marshes often act as refuges for many bird species when water levels in the salt marsh are high.

Wetland-upland transition or ecotone

The transition zone surrounding coastal wetlands is where both wetland and upland vegetation overlap at elevations between 7.5 and 9.8 ft MLLW (1.4 and 2.1 m NGVD, where NGVD = National Geodetic Vertical Datum; Figure 2.26). Salt marsh plant species typifying this habitat include alkali heath (*Frankenia salina*), sea lavender (*Limonium californicum*), perennial glasswort (*Salicornia subterminalis*), salt-grass (*Distichlis spicata*), and Watson's saltbush (*Atriplex watsonii*). Upland species typifying this habitat include natives such as box-thorn (*Lycium californicum*), yerba reuma (*Frankenia palmeri*), California saltbush (*Atriplex californica*), and pineapple weed (*Ambylopappus pusillus*) (see Zedler and Cox 1985, James and Zedler 2000) (Section 2.3.3). Salinities in this habitat are highly variable, depending on rainfall and extreme flooding events. The dry and saline nature of this habitat makes it inhospitable to most perennial exotic species. The abundant invaders are Australian saltbush (*Atriplex semibaccata*), crystal iceplant (*Mesymbryanthemum crystallinum*), and the sow thistles (*Sonchus asper* and *S. oleraceus*) (Ferren 1985, Cox and Zedler 1986). Many exotic annual species can complete their short life cycles in such sites during brief periods of high moisture and low salinity. Examples are little ice plant (*Mesymbryanthemum nodilforum*), European sicklegrass (*Parapholis incurva*), and rabbitfoot beardgrass (*Polypogon monspeliensis*). Many bird, mammal and reptile species can be found in these areas (see Zedler et al. 1992). Reptile species include the California kingsnake (*Lampropeltis getulus californiae*), San Diego gopher snake (*Pituophis melanoleucus annectans*) and the side blotched lizard (*Uta stansburiana*) (Zedler et al. 1992). Mammal species found here include the western harvest mouse (*Rheithrodontomys megalotis*), the California jackrabbit (*Lepus californicus*), and the coyote (*Canis latrans*).

Plates depicting the most common species of plants are included in the middle of this book. More information on the plants of southern California coastal wetlands can be found in Ferren (1985), and more detail on the diversity of wetland habitats appears in Ferren et al. (1995). The latter document includes a detailed habitat classification system for the region's coastal wetland habitats.

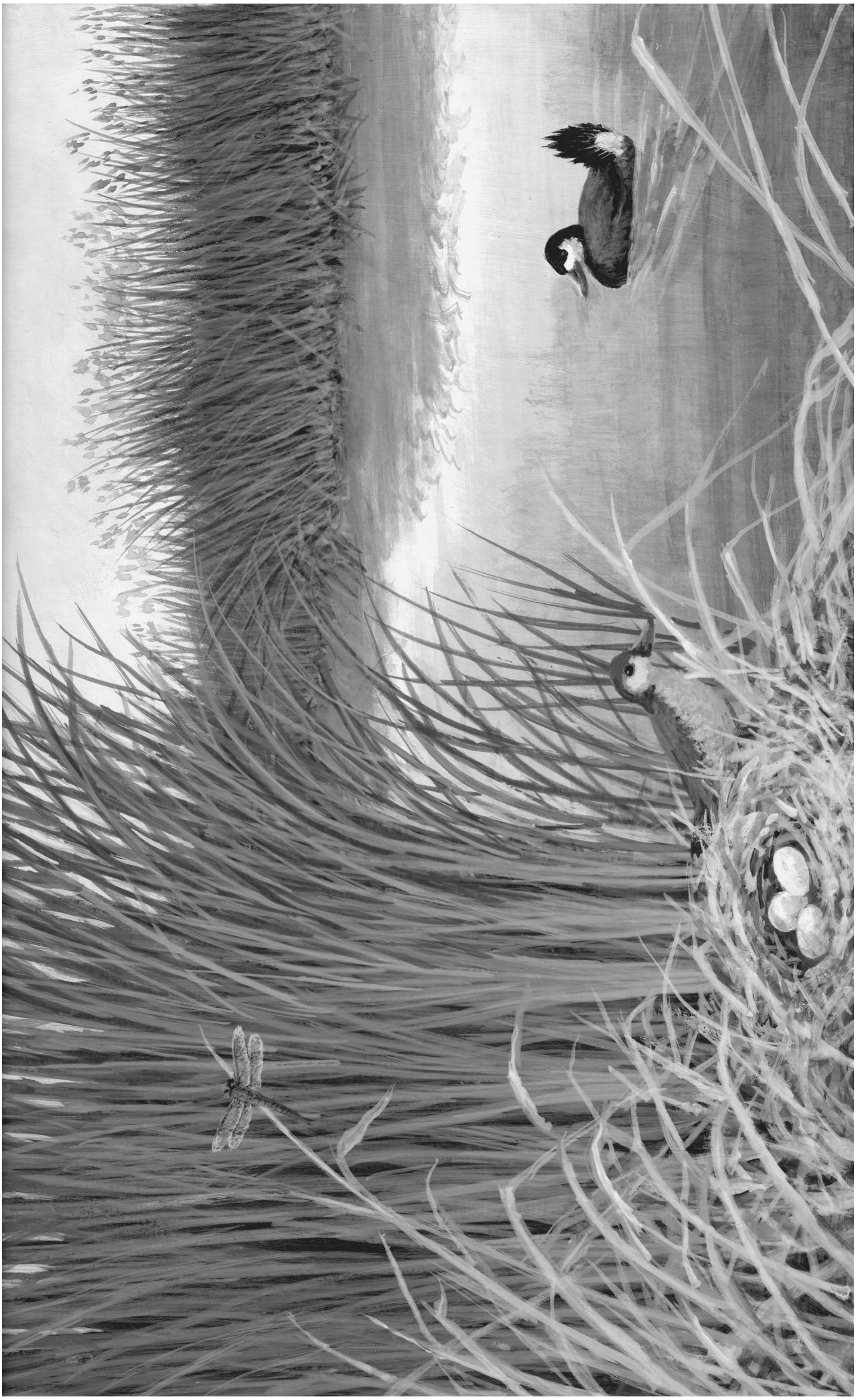

***Figure* 2.25** Brackish marsh habitat with cattail and ruddy duck in the foreground and bulrush in the background. McIntire Drawings, © 1999 by Zedler.

Figure 2.26 Upland transition habitat, with a ground squirrel and the woody box-thorn (on the right), high marsh in the foreground, including the endangered salt marsh bird's-beak (lower right) and coastal scrub vegetation in the background. McIntire Drawings, © 1999 by Zedler.

Box 2.6 Restoring endangered plant populations

Meghan Q. Fellows

"Unintelligent tinkering with endangered plants is extremely risky, because most species are critically endangered by the time they are federally listed" (Berg 1996). Although many restoration plans call for endangered species to be reintroduced, the low success (under 30%) of such efforts indicates the need for better planning and implementation (Hall 1987, Allen 1993). Each endangered species has its own characteristics, and each needs individual guidelines for restoration. Hence, restoration techniques for endangered plants are inherently "experimental" (Allen 1993). It helps to know why the species has become rare (cf. Rabinowitz 1980). With answers to several questions, plans can be developed to reintroduce extirpated species or enhance declining populations.

The issue of rarity

All endangered plant species are rare, but not all rare plant species are endangered. According to Rabinowitz (1980), there are "seven forms of rarity," generally meaning a limited geographic distribution, habitat preference or local abundance. However, some species can be locally abundant while restricted both geographically and ecologically. Long-term records on abundance help distinguish naturally rare species from those that are endangered, i.e., those that are noticeably declining. Some plant species benefit from sparsity by having reduced herbivore pressure and less intraspecific competition (ibid.). If local sparsity is important, such species should be restored in population numbers consistent with the least disturbed populations (Howald 1996, Mladenoff et al. 1997).

Questions to be answered to aid planning

Has there been adequate scientific study of the species and the restoration site? The biology of most endangered species is poorly known. Without a thorough understanding of the ecological and reproductive requirements of the endangered species, reintroductions are unlikely to succeed (Falk et al. 1996, Howald 1996). Aspects that need to be known include the minimum viable population size, critical pollinators, mycorrhizal symbionts, seed dispersal agents, and nutrient and hydrologic requirements (Allen 1993).

Is quality habitat available within the species' geographic range? The reintroduction range should consider the former range and the species' potential range following global climate change (Falk et al. 1996, Falk and Olwell 1992). The lack of appropriate and available habitat may limit reintroduction attempts. Falk and others (1996) recommend that mitigation sites not be used for reintroductions because of the high chance of failure. At the same time, reintroduction into a rare native ecosystem may cause irreversible damage. For example, vernal pools are a threatened wetland type and their mima mound topography is also rare, but new vernal pools are often created among natural pools with seeding from external sources. This type of restoration might put both the natural vernal pools and the restored pools at risk, because both have unnaturally high densities in the "restored" landscape. The carrying capacity of the natural or restored site may prevent development of a sustainable population of the endangered species (Howald 1996, Mladenoff et al. 1997).

Is adequate source material available of the appropriate genetic variability? As suggested by Lacy (1994), diversity is needed at the global scale, while similarity to local gene pools is needed at the landscape scale. Removing propagules from a rare population may reduce its chances for survival. Seed sources may thus be limiting to restoration efforts. In southern California, *Cordylanthus maritimus* ssp. *maritimus* (salt marsh bird's-beak) can be collected at a rate of only 500 seeds per population, despite its low genetic diversity (Helenurm and Parsons 1997). Other species may have lower collection limits, especially if the species has been reduced to one individual or population. Of the 332 taxa listed as endangered by the U.S. Fish and Wildlife Service between 1985 and 1991, the median total population size was 120 individuals (Berg 1996).

Questions to be answered to aid implementation

Is there cooperation between biologists, agency representatives, and all interested parties involved? It is important that the restoration program have broad support (Hall 1987, Allen 1993). If the restored habitat does not receive long-term protection or the reestablished populations are in danger of being destroyed, the site is unsuitable for endangered species reintroduction. A reintroduced population of Mead's milkweed (*Asclepias meadii*) was allegedly destroyed by biologists who disagreed with introducing nonnative genetic material into the area (Allen 1993). In Southern California, habitats may be unintentionally disturbed by recreational use, including off-road vehicles, which may prevent sustainable populations of endangered species from developing. The Yellowstone wolf reintroduction may fail if ranchers remove the wolves from the area. If careful, long-term preservation of the site *and* the reestablished population is assured, the time, effort, money, and propagation materials used in the attempted restoration will not be wasted.

Is there adequate time to allow for restoration to occur at the optimal, ecological time? For some species, planting or transplanting should occur only at certain times of the year. The species' ecological requirements need to be identified and used for timing of reintroduction. If plants are transplanted in the middle of flowering, for instance, the reproductive output would be lost from both the donor population and the restoration site. Reintroducing at an appropriate time of year will cause less damage to both the donor and restored population.

Has there been adequate planning? A model science-based restoration planning effort is that of Pavlik et al. (1993), who attempted experimental restoration of *Amsinckia grandiflora*. First came a detailed site selection process, which sought to optimize both ecological (macroclimate, soil, slope, exposure, community associates, habitat size, degree of disturbance) and logistic factors (land use history, road access, property ownership, size). Second, seed material was obtained from both wild and cultivated sources (originally from the wild). Germination trials and a genetic study were completed to ensure that they were working with a viable, genetically diverse donor seed source before the reestablishment occurred in the field. The restoration plan included adequate site preparation and long-term maintenance, with both herbivore and exotic weed exclusion. Planting was timed to coincide with natural rainfall. Germination and reproduction were extensively monitored to ensure the project could be judged on a scientific basis. Finally, after the experimental restoration was completed, the authors published the results and disseminated the information to a broad range of concerned audiences.

Has the plan been peer reviewed? Peer review of plans by parties not personally involved is very useful (Hall 1987, Howald 1987). Allowing scientists, managers, and naturalists to evaluate the restoration plan will help ensure that essential information is included.

Questions to be answered to ensure proper follow-up

Has a system of monitoring and adaptive management and maintenance been established? Compliance criteria and standards differ for each mitigation project. A bond may be held by the government until the mitigation project has met the required performance standards. Hall (1987) reported that in some cases, bonds were returned to the mitigating party as soon as plants were placed in the ground, allowing no time to determine if the desired population established or was self-sustainable. Mitigation agreements usually require that a population reach a certain size and has a certain distribution over the habitat area for a certain amount of time. In 1987, the average monitoring period ranged from 3 months to 1 year (Hall 1987); more recently, such periods average 3 to 5 years (Mitsch and Wilson 1996). Most restorationists agree that even 5 years is too brief to assess the long-term self-sustainability of a population (ibid.). "Reintroduction," says Allen (1993), "can only be considered complete when a species is safely reestablished in its ecological and evolutionary context."

Are there plans to disseminate the results of the reintroduction attempt to the public, agencies, and the corporations involved? Improvements in the ability to restore endangered species depend on information sharing (Howald 1987). There is rarely enough time to gain the knowledge required for a successful reintroduction project before the initiation of the project.

Conclusion

We never know enough to ensure that a reintroduction effort will result in a self-sustaining population. Hence, we strongly recommend a phased approach, with experimentation prior to full-scale implementation. Pilot efforts conducted in an experimental fashion will yield more information than trial-and-error approaches. Our efforts to reestablish an endangered salt marsh plant (Box 2.7) and enhance declining species (Box 2.8) are provided as examples of how research and restoration can proceed in an adaptive management setting.

Box 2.7 Restoring salt marsh bird's-beak to San Diego Bay

Meghan Q. Fellows

The reintroduction of salt marsh bird's-beak (*Cordylanthus maritimus* ssp. *maritimus*) to Sweetwater Marsh (Box 1.6) offers an example of how field experimentation and an adaptive management approach can improve restoration efforts. This plant is both a state- and federally-listed endangered species.

Salt marsh bird's-beak has a locally abundant population over a narrow geographic distribution along the coast of Baja California, Mexico, to Point Morro, California, and it occupies a narrow habitat type, the high salt marsh (Appendix 2, Hickman 1993). Its population has declined along with the loss of coastal wetland habitat to development. The last naturally occurring plants in Sweetwater Marsh were seen in 1988. This species is an annual, so it must produce seeds in most years to persist, and it is a hemiparasite,

so it needs suitable host species to grow to maturity. Germination requirements (lowered salinity) and low tolerance to inundation confine it to areas that are infrequently inundated by tides, while its low stature (often under 15 cm in height) and moisture and host requirements appear to limit its growth in uplands.

Where the California Department of Transportation (Caltrans) filled habitat within Sweetwater Marsh (Box 1.6) that could potentially have supported salt marsh bird's-beak, the U.S. Fish and Wildlife Service required that suitable habitat be created (by excavating fill) and that a self-sustainable population be planted from seed. The Pacific Estuarine Research Laboratory (PERL) directed the reintroduction, studied the results of the restoration attempt (Parsons and Zedler 1997, Helenurm and Parsons 1997, unpublished PERL monitoring reports to Caltrans), and developed the project into an adaptive management program (J. Zedler, *personal communication*).

The nearest natural population at Tijuana Estuary (Box 1.9) consists of about 100,000 individuals, thus it seemed unlikely to be negatively impacted by the removal of seeds (*personal observation*). Genetic studies were not conducted until after the reestablishment attempt. We know now that the donor population appeared to be as diverse as other populations in the habitat range (Helenurm and Parsons 1997). Studies on habitat preference (cover requirements and salt effects) were conducted prior to the reestablishment attempt (Fink and Zedler 1989*a,b*).

Brian Fink conducted the reestablishment effort in conjunction with PERL. The first year, he seeded a small remnant of high marsh that had been made into a wetland island by excavating fill to create channels and lower marsh habitat. While seedlings grew and matured, few seeds were produced. He hypothesized that pollinators were limiting, as the small island lacked upland areas for the preferred pollinators to nest (pollinator limitation was later demonstrated by Parsons and Zedler 1997). The project became an example of adaptive management when this information was brought to the Highway Department and the Fish and Widlife Service, along with a recommendation to move the reintroduction effort to the larger high marsh remnant known as Sweetwater Marsh. The agencies agreed, and seeds were subsequently sown for three years before compliance monitoring began. Utilizing this higher quality ecosystem to support a reintroduced population increased the chance of seed production, as the salt marsh vegetation was adjacent to upland and there were more patches of the species' preferred host, *Monanthochloe littoralis.*

Salt marsh bird's-beak seeds were originally sown in small patches that were mapped and flagged. Plants were then monitored by searching the area and counting all individuals (or estimating where they became too dense for counting). Later, after the population expanded well beyond these patches, including the Connector Marsh (Box 1.7) and Marisma de Nación (Box 1.8), a global positioning system was used to mark the perimeters of patches, and plant numbers were estimated within patches. The population grew from about 5000 plants in the first year of compliance monitoring (after seeding had ceased) to about 14,000 plants in the subsequent two years. At that point (1995), the mitigation requirements (at least 100 plants in 5 patches at least 10 m apart, persisting or expanding for 3 years) had been met.

The next year, 1996, was a dry year, and the population declined and shrunk to a small area of lower topography (perhaps a "moisture refuge"). Subsequently, plants have reappeared in some of the former patches within the constructed wetlands, and a large population has developed where channel excavation allowed a tidal creek to erode into a salt panne. Presumably, the improved tidal flows lowered soil salinities and allowed salt marsh bird's-beak to grow and reproduce.

Long-term prospects for protecting this habitat are high; the salt marsh is a U.S. Fish and Wildlife Service National Wildlife Refuge (Box 1.6). Biologists at the Highway Department

and at the Fish and Wildlife Service have had a good working relationship. Local nature interpretive centers provide information to the public, and many talks and articles and books on local wetlands and their endangered species have been made available to both scientists and the public (Zedler 1996*a*, Zedler 1998). Two recent master's theses and a doctoral dissertation have documented information on this species at the reestablishment site, i.e., its pollination and seed production (Parsons 1994), the host-parasite interactions (Fellows 1999), and its germination and establishment (Noe 1999). Future publications of these findings will make this information more widely available.

The reestablishment of salt marsh bird's-beak involved the application of scientific knowledge, the cooperation of scientists and public agencies, and long-term protection of the habitat. Although prospects for long-term persistence are high, extirpation is always a possibility. It is still not known why the original population died out. Monitoring is no longer funded, and research efforts have ended. There is still a need to determine the size and longevity of the seed bank. The highest chance of success for this establishment attempt was ensured by the careful consideration of all factors, ecological, political, and social, associated with endangered plant species restoration.

Box 2.8 Enhancing rare plant populations at Tijuana Estuary

Gabrielle Vivian-Smith and Joy B. Zedler

In 1984, two short-lived species (*Salicornia bigelovii, Suaeda esteroa*) experienced major population declines when the mouth of Tijuana Estuary closed and marsh soils became dry and hypersaline during the ~8-month nontidal period (early April to mid December; Zedler et al. 1992). These population declines drew alarm because both species were lacking in most of the region's wetlands (Appendix 3), presumably because of similar tidal closures. In the 15 years following tidal restoration, neither species reestablished its historical distribution at Tijuana Estuary (Box 1.9).

Both species rely upon a transient seed bank, with high densities of seedlings present in late winter 1984, but virtually all plants dying during closure. Any remaining viable seeds probably failed to germinate due to physical stresses. The following spring, only a few plants were seen in areas near freshwater inflows.

One hypothesis for why these species failed to reestablish is the local loss of their seed banks. Alternatively, they may have failed to reestablish because canopy gaps were lacking for germination and establishment. The opportunistic growth of other species (notably *Salicornia virginica*) during the tidal closure period might have increased canopy cover and decreased rooting space to levels unsuitable for seedling recruitment.

We tested these alternative hypotheses by establishing a reintroduction experiment at Tijuana Estuary during 1998. We chose areas where *Suaeda esteroa* and *Salicornia bigelovii* occurred in high numbers prior to 1984. The experiment had separate seed-addition treatments (+*Suaeda esteroa* and +*Salicornia bigelovii* seeds) and controls with no seeds added. At the same time, we imposed two canopy treatments, removal of all species and removal of just *Salicornia virginica.* The plots were circular and nested, with 0.25-m^2 plots for canopy removal and 0.10-m^2 for seed addition; plots were distributed equally across five subareas (blocks) within the tidal marsh. Additional bags of seeds were placed on the

marsh surface to assess field germination rates. Reestablishment of *S. esteroa* and *S. bigelovii* was then monitored through October 1998.

Both the seed bank and plant canopies limited reestablishment. Seedlings appeared only where seeds were added (all control plots had 0 seedlings). By October 1998, *Suaeda esteroa* had recruited only 2 plants out of 2000 seeds added, even though its field germination rate had been 46%. In contrast, 54% of the *Salicornia bigelovii* germinated in bags but only 453 seedlings appeared in experimental plots (23% of the 2000 seeds added). Germination was thus followed by high seedling mortality, which was recorded during the study. Seedling establishment increased with larger amounts of the canopy removed (14% where no vegetation was clipped, 27% with just the *Salicornia virginica* canopy removed, and 36% where all vegetation was removed).

The difficulty of reestablishing rare species is illustrated by this pilot project. One species had little recruitment and the other produced low densities relative to historical levels (thousands per m^2), especially where the marsh canopy was left intact. The importance of conducting reintroductions as experiments is also clear. We demonstrated the importance of opening the canopy to facilitate seedling establishment. Keeping the canopy open remains a problem, as plant cover was high (>40%) by October 1998 in plots where either some or all vegetation had been removed prior to seed addition.

Further experimental research will seek to understand why the canopy was able to remain open prior to 1984, why canopy cover is so much higher at present, and whether other restoration measures can increase the rate of reestablishment for these regionally rare salt marsh plants.

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chapter three

Hydrology and substrate

John C. Callaway

3.1 Introduction

Hydrology is the driving force for wetland development and functioning (Mitsch and Gosselink 1993). In discussing hydrology here and elsewhere in the book, we use the term "hydrology" in a very broad sense, covering all aspects of water flow and its impacts on a restored wetland, including geomorphology, sediment transport, water and soil salinity, and inundation regime. Hydrology affects soil development, sediment dynamics, plant growth and dispersal, aquatic animal access, and many other processes. For any wetland restoration to provide habitat support and other functions, appropriate hydrology must be established, and this frequently means more than simply restoring tidal action by removing berms or levees. Together, hydrology and substrate conditions (including salinity, texture, organic matter content, and nutrient status) are the key abiotic factors that influence the development of wetland plant and animal distributions.

Salt marsh restoration attempts in southern California and elsewhere often start with grading the site to proper elevations, restoring tidal access, and planting vegetation (Figure 3.1). While these efforts may allow initial establishment of wetland vegetation, many restored sites have not functioned equivalently to natural wetlands (Haltiner et al. 1997, Zedler and Callaway *in press*). In many cases, hydrology and soils at the restored sites differ from natural salt marshes, and improper hydrologic and soil conditions have contributed significantly to the poor functioning of restored sites (Craft et al. 1988, Langis et al. 1991). In planning, implementing, and evaluating restored wetlands, hydrology and substrate should be considered together. Wetland hydrology determines conditions for sediment accumulation and/or erosion, which will affect substrate conditions at a restored site. Furthermore, the degree of localized flooding within a wetland can affect rates of organic matter accumulation, as well as other soil processes, such as nutrient dynamics, redox chemistry, and salt accumulation.

Under natural conditions, tidal wetlands are primarily depositional systems, characterized by low velocity flow in tidal channels and over the marsh. This results in the gradual accumulation of finely textured, highly organic sediments. The complex network of tidal channels delivers sediment and nutrients to the wetland surface and provides many other important functions, including access for organisms and flushing of salts or other material that can accumulate in wetland soils. Given the link between hydrology and substrate processes, we need to consider the restoration of a particular wetland in its geomorphic framework. Intertidal wetlands are a function of their tidal hydrology, freshwater inflows,

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Figure 3.1 Excavation of a tidal channel and the creation of the Tidal Linkage at Tijuana Estuary (see Box 1.10 for details). Material was excavated to create elevations ranging from subtidal to high marsh, with the majority of the area at mid-marsh plain elevations.

sediment inputs, sea-level rise, subsidence, storm impacts, and other extreme events. Finally, the relationship of a restoration site to existing geomorphology of the estuary, coast, and watershed should be considered in designing a restored wetland. Hydrologic processes outside of the restored wetland are also likely to affect the site. Bedford (1996) evaluated issues related to the hydrologic restoration of freshwater wetlands and concluded that a broad, landscape approach is necessary. These same landscape considerations apply to tidal wetlands and other habitats (Bell et al. 1997).

3.1.1 Restoration and creation

Coastal wetland projects cover a broad spectrum from restoring minimally impacted sites to creating new wetlands (Section 1.4). Restoration of former wetland areas requires less manipulation than the creation of wetlands from uplands or subtidal habitats, and restoration projects also have a higher likelihood of achieving goals (Kruczynski 1990). The degree of alteration or degradation at a restoration site determines where it falls along this spectrum (see Table 1.2) Sites that are only minimally degraded are likely to retain appropriate soil conditions, elevations, and topographic complexity, so that grading would not be necessary before tidal action is restored. These sites will make more rapid progress toward the restoration target. For example, in the Salmon Estuary in Oregon, Frenkel and Morlan (1991) found that a diked pasture quickly developed a typical Pacific Northwest salt marsh community after dikes were breached, although the original high salt marsh community was not restored.

Restoration sites that have undergone extensive modifications may need grading and other manipulations to restore appropriate wetland conditions. For example, in the Sacramento-San Joaquin Delta, many hydrologically altered wetlands have undergone severe subsidence due to the oxidation of organic soils following drainage of these wetlands (Rojstaczer and Deverel 1995). In order to restore areas such as this, elevations must be increased; otherwise, hydrologic control structures are needed to maintain proper water levels (e.g., muted tidal conditions at the Bolsa Chica Ecological Reserve in Long Beach, California). Hydrologic control structures have been used extensively in managed wetlands

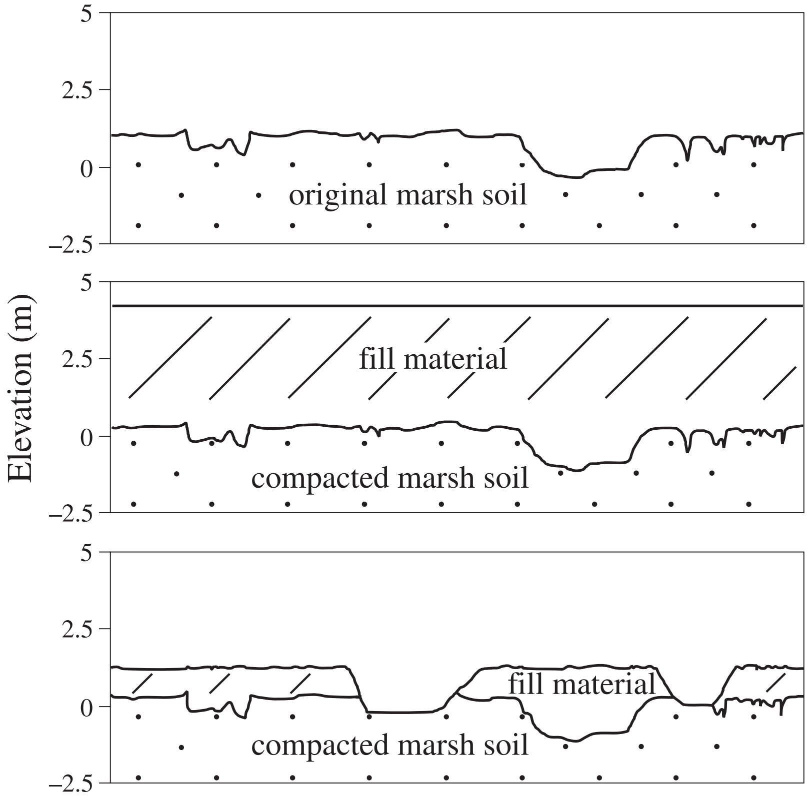

Figure 3.2 Cross section of a "filled" wetland with resulting compaction and lowering of elevation. Top panel shows conditions at a natural wetland prior to filling; middle panel illustrates the same site following filling and compaction of the natural wetland soil; bottom panel illustrates the topography and substrate following restoration to elevations similar to original wetland.

in the Mississippi River Deltaic Plain of Louisiana, although with mixed results (Boesch et al. 1994). Without some hydrologic manipulations, restoring tidal flow to an area that has subsided would create a subtidal pond, rather than an intertidal wetland. If a former wetland has been extensively filled, the original wetland soil may be so compacted that removal of fill would leave the wetland soil surface substantially below the "project elevation" for intertidal flooding (Figure 3.2). The original soil would be best for restoration, but it would be buried below fill, unless exceptional measures were taken to overexcavate the site, stockpile the original wetland soil, and regrade the area to the correct elevations. In addition, restoration sites that have been graded, cultivated, or diked may no longer retain the topography or qualities of natural wetland systems. Soil texture may be suitable, but soil organic matter may be lacking, due to enhanced decomposition under aerated conditions.

Creating a wetland from upland requires the greatest effort in the design and grading phases because the hydrology, soil, and other physical parameters will not match those of natural wetlands. Hence, the outcome of a created wetland is the least predictable. The goal is still a functional wetland; however, it is more difficult to guarantee the exact form of the finished product. Wetlands are very dynamic, and the best designs encourage processes that enable the system to develop itself, rather than trying to duplicate an existing natural marsh (Mitsch and Wilson 1996) (Section 4.2). The focus of this chapter is to review hydrologic concerns at coastal wetland restoration sites and provide a more detailed review of soil considerations and potential problems.

3.2 Hydrology

3.2.1 General hydrologic considerations

Because hydrology drives the geomorphic development and functioning of a restored wetland, the establishment of appropriate hydrologic conditions is one of the first considerations in the design of restoration projects. For example, it is essential that tidal creeks be designed large enough to allow adequate tidal flushing of the restored wetland. The configuration (size and cross section) of creeks should be based on natural creek morphology.

Initial attempts at coastal wetland restoration focused on the basic hydrologic concepts which affect the development of a restoration site, including:

- the proper size for a tidal inlet,
- the tidal prism necessary to maintain flows and keep a tidal inlet open,
- the degree of scouring, given a particular flow and sediment grain size, and
- the elevation necessary for a particular frequency of tidal inundation.

Most of these basic considerations have been evaluated thoroughly, and hydrologic models have been developed to guide basic hydrologic design of restored wetlands (Williams 1996, Coats et al. 1995, Haltiner et al. 1997, Williams *in press*). However, in some cases the restoration of tidal action at a site is difficult, and it is constrained by factors such as adjacent land use and existing infrastructure (Chapter 2). Furthermore, providing appropriate hydrology is not as simple as establishing the proper elevations at a site.

In order to improve on hydrologic restoration, we need to consider the details of:

- the distribution and size of tidal creeks across a wetland and within particular habitat types,
- creek sinuosity,
- creek stability and migration, and
- creek morphology (e.g., how to create small, steep-sided creeks).

Many of these "details" are critical for plant and animal distributions and for biotic processes; however, we have little information from natural wetlands to guide us in the design of these features. In order to restore the full range of natural functions, these issues need further consideration and evaluation.

Most wetland restoration sites lack the hydrologic complexity of natural wetlands (Section 5.2 and Figure 5.5). For example, deep subtidal basins have been created to provide subtidal fish habitat at various wetland sites (Box 1.2, Box 1.7) (Zedler et al. 1997). These deep basins lack the gradual, sloping interface between subtidal and intertidal habitats, and they lack the natural range of wetland habitats. Thus, they are unlikely to provide the essential refuges and spawning habitats required by the various fish species found in natural coastal wetlands. Other coastal wetland restoration sites have been designed with a single large channel (Box 1.8). The hydrology of a single, large tidal creek does not replicate the topographic heterogeneity and drainage patterns found in a natural wetland. Recent research in southern California (Desmond et al. 2000, West and Zedler *in press*) and in other areas (Weinstein 1979, Rozas and Odum 1988, Hettler 1989) has documented that first- and second-order creeks provide important functions within tidal wetlands, including foraging and nursery support for fish and invertebrates (Sections 2.4 and 5.3).

In grading restored wetlands, it is much easier for bulldozer operators to construct smooth surfaces and edges, but these features are not natural (Adam 1990, Chapter 2).

Variations in elevation and flow associated with complex topography are important to create areas of differential sediment deposition and erosion, which will further increase topographic heterogeneity. Large, straight tidal channels are relatively easy to construct and are a typical component of most restored wetlands, while small, stable creeks are difficult to create in a tidal environment and frequently are lacking in restored wetlands (Figure 3.3). The construction of intricate tidal creek networks in restoration sites is a challenge that leads to extra costs. Further research to improve engineering and construction methods for these creeks will be valuable, as will evaluation of the effects of creek networks on the development of wetland function (Box 1.11). It may be difficult to create stable, small creeks in a tidal environment.

3.2.2 Processes of natural wetland development

Most coastal wetlands are thousands of years old (Chapman 1960, Mitsch and Gosselink 1993), and their natural development has taken place over a scale of centuries (Redfield 1972). In restoration, we attempt to condense the time required to achieve a fully functioning marsh. Hence, we need to understand the processes that contribute to natural marsh development and identify controlling mechanisms.

Intertidal coastal wetlands are found in areas protected from direct wave impacts. They develop behind bars, islands, or spits, in large protected bays, and in deltaic systems (Chapman 1960). In order for marshes to develop, there must be a relative balance between sediment accretion and relative sea-level rise, which is equal to eustatic sea-level rise plus local changes due to subsidence or tectonic processes. Eustatic or global sea-level rise is caused primarily by thermal expansion of the ocean, melting of glaciers and ice sheets, and is estimated to be 10 to 25 cm over the last 100 years (Warrick et al. 1996); rates of relative sea-level rise may be an order of magnitude greater than this in areas of significant subsidence or tectonic activity. Under conditions of relative balance in these factors, intertidal salt marshes tend to develop a flat marsh plain near mean high water (MHW) (Myrick and Leopold 1963, Pestrong 1965, Redfield 1972, Zedler et al. 1999).

Intertidal wetlands exist as part of a continuum of habitats within coastal and estuarine systems (Box 2.5). This range of habitats includes subtidal areas, intertidal flats, tidal creeks and channels, salt marsh, salt panne, brackish marsh, and wetland-upland ecotones. These habitats develop due to a combination of geomorphic processes (sea-level rise, sediment inputs, protection from wave energy, scour of tidal flow, etc.), and each habitat type is a reflection of the balance of these processes at a given location within the estuary. Changes in the intensity of these factors will result in habitat shifts. In addition, impacts to one habitat type are likely to affect adjacent habitats, i.e., loss of mudflats will affect the adjacent vegetated marsh through changes in wave protection and other factors. One conceptual model of how natural tidal wetlands develop shows upland areas being flooded as sea level rises; another depicts the gradual accumulation of sediment and conversion of subtidal areas to intertidal mudflats and, eventually, vegetated wetlands (Mitsch and Gosselink 1993). The applicability of these two models depends on environmental conditions at a given location.

In most natural marshes, sedimentation rates are in equilibrium with relative sea-level rise, resulting in a stable elevation of the marsh plain. Rates of sedimentation are greater in areas that receive more frequent tidal flooding (i.e., lower elevations and tidal creek edges), resulting in a feedback that causes the marsh-plain to stabilize at elevations between MHW and MHHW (mean higher high water) (Allen 1990, Pethick 1992, Allen 1994). The relative elevation of the marsh surface is affected by many factors (Figure 3.4). In addition to the feedback between elevation and sediment inputs (both tidal and storm inputs), other factors are also linked to elevation via a feedback mechanism, including

Figure 3.3 **(A)** Large, straight tidal creeks (such as this one from Muzzi Marsh, San Francisco Bay) are commonly created at most restoration projects. **(B)** Smaller tidal creeks may take a long time to develop and are frequently lacking at restored wetlands.

biomass production (above- and belowground) and decomposition. Eustatic sea-level rise, subsidence, and tectonic activity affect the relative elevation of the marsh, but without any feedback mechanism.

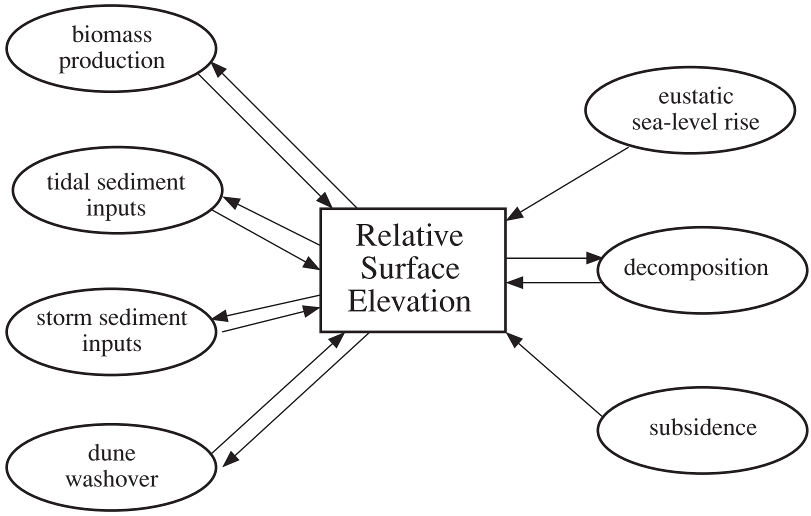

Figure 3.4 Feedback processes that affect the relative elevation of the marsh surface. Factors on the right side of the diagram cause decreases in the relative surface elevation; factors on the left side lead to increases in relative surface elevation. Tectonic activity (not shown) also contributes to relative surface elevation and can cause both increases or decreases, depending on the particular event.

As an intertidal wetland develops, a balance is reached between the tidal prism (the volume of water that flows in and out of the system on a tidal cycle) and the geometry of the tidal channels that accommodate this flow (Williams 1986, Coats et al. 1989). This hydrogeomorphic balance is similar to that for a river channel and its flood plain, where erosion and sedimentation are controlled by river flow and sediment load (Leopold 1997). In the case of coastal wetlands, the elevation of the marsh plain will vary with the tidal regime (mixed vs. even tides), as well as the availability of sediment (Frey and Basan 1985).

In many cases, the tidal prism of the wetland and/or estuary has been reduced by anthropogenically enhanced sediment inputs. In southern California, Onuf (1987) documented a 40% reduction in the tidal prism of Mugu Lagoon as major floods brought in sediments from agricultural fields within the watershed. Similarly, many lagoons in San Diego County have filled more rapidly than under natural conditions due to urbanization of their watersheds. Sedimentation has serious consequences for wetland hydrology, especially in small coastal lagoons. A reduced tidal prism decreases the velocity of tidal water that flows through the lagoon, especially at the tidal inlet. Slower currents allow more sediment deposition and further reductions in channel cross-sectional area. These processes result in the progressive reduction of the tidal prism, eventually leading to the closure of the tidal inlet. Hence, restoration and maintenance of the tidal prism is an important consideration for coastal wetland restoration projects.

In addition to sediment deposition and erosion processes, soil organic matter accumulation is also significant in the development of the marsh plain. Tidal wetlands tend to have relatively high organic matter content, due to greater rates of organic matter production relative to decomposition. In many cases, organic matter accumulation is the most significant factor affecting overall accretion rates (Bricker-Urso et al. 1989, Nyman et al. 1990, Craft et al. 1993, Callaway et al. 1997). In order for a restored wetland to develop naturally, conditions must be suitable for the ongoing production and accumulation of organic matter (Section 3.3.2).

3.2.3 Formation and characteristics of tidal creeks and channels

The marsh plain is dissected by a system of creeks of various orders and sizes (Box 2.1). These tidal creeks connect the vegetated marsh and mudflats with the adjacent lagoon or estuary, providing inputs of sediments and nutrients and flushing salts and other materials from intertidal areas. When tides flood the marsh surface, water flow slows in channels, and overbank tides flow as a sheet across the marsh at even slower velocities (Wang 1997). Slow flows allow the deposition of mostly fine sediments, with coarser grained material dropping out in deeper channels or on mudflats adjacent to deeper water (Pestrong 1965).

As the site builds in elevation, creeks form to facilitate drainage. Creeks develop through a variety of processes: headwater retreat, downward incision, deepening of channels, and lateral migration (French and Stoddart 1992). Marsh creeks tend to retain characteristics of intertidal mudflat creeks, although the former are more stable laterally where vegetation inhibits their migration (Collins et al. 1987). Over time, tidal flows tend to develop creeks that are highly sinuous (Pestrong 1965), providing increased edge adjacent to the tidal creeks and direct drainage of large areas. In natural wetlands, roots and rhizomes hold creek banks in place, leading to the development of steep, narrow channels throughout the salt marsh. Channels in adjacent mudflats tend to be shallower and wider (Frey and Basan 1985, Eisma 1997). Creek stability is an important issue for restored wetlands, and it remains to be demonstrated how we can create small, steep-banked creeks that are hydrologically stable before the vegetation is fully restored.

Creek characteristics (depth, slope, order, etc.) at a particular location are a function of the area of wetland and the local tidal regime (Haltiner et al. 1997). The configuration of tidal channels can vary greatly across wetlands, depending on tidal regime, sediment input, sediment type, vegetation, and other factors. Zeff (1999) found that small, dead-end creeks are more sinuous than larger, through-flowing channels in a New Jersey salt marsh. Pethick (1992) and Eisma (1997) have both classified tidal creeks, and they consider the following factors in their classification: sinuosity (straight, sinuous, or meandering), depth, slope of bank, creek order, and adjacent vegetation.

In addition to individual creek characteristics, the network of creeks at a site also has specific features. Networks have been described as parallel, meandering, dendritic, elongated dendritic, distributary, braided, and interconnecting (Eisma 1997; Figure 3.5). How these systems fit together will determine the drainage density of a wetland (number of creeks/unit area), the distance from a location within the wetland to an adjacent creek, and other factors that affect biological processes at the wetland (Section 5.2.5).

To date most created and restored wetlands have been too simplistic in their creek morphology and topography. Common problems with creek networks at restoration sites include:

- low density of creeks,
- no small, first-order creeks,
- lack of creek sinuosity,
- no steep-sloped banks; lack of creek bank stability,
- emphasis on subtidal creek habitat rather than intertidal creeks, and
- improper topography along the interface between creek bank and marsh to facilitate overbank flooding and small-scale heterogeneity.

Besides lacking small creeks, restored sites also lack small-scale features such as pannes and other topographic variability that provides important complexity to natural sites. Because of the strong link between tidal creek hydrology and biological processes, differences in tidal creeks will have implications throughout the ecosystem, including impacts

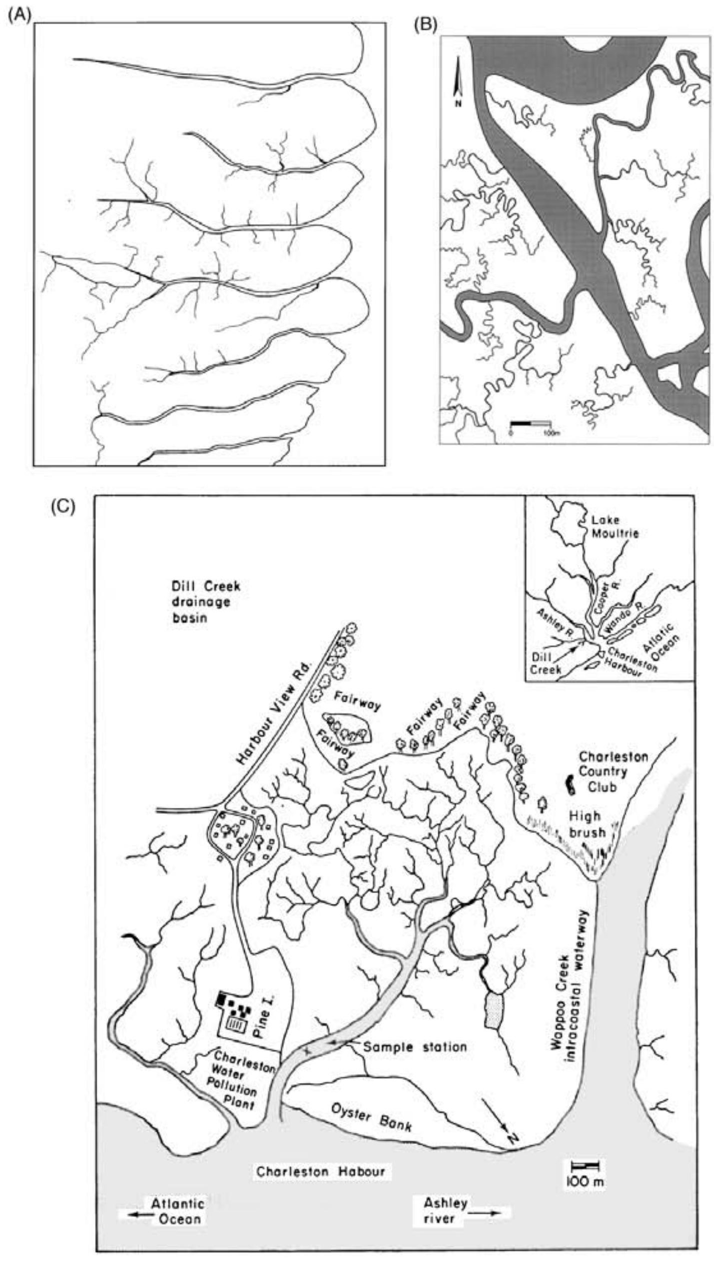

Figure 3.5 Some channel systems found in natural wetlands. A: parallel; B: meandering; C: dendritic. (Reprinted with permission from D. Eisma, 1995, Intertidal deposits: river mouths, tidal flats, and coastal lagoons. Lewis Publishers, an imprint of CRC Press, Boca Raton, FL).

on soil biogeochemistry (King et al. 1982, Howes and Goehringer 1994), plant distributions (Zedler et al. 1999), and fish and invertebrate establishment and use (Moy and Levin 1991, Minello et al. 1994, Minello and Webb 1997).

In order to create more natural tidal hydrology for restored wetlands, characteristics of creek systems from nearby natural reference wetlands should be used as a model. At Tijuana Estuary, Desmond (1996) found that 1st-order creeks make up 45% of the total creek network. Pestrong (1965) measured similar parameters for salt marshes in south San

Francisco Bay. Zeff (1999) summarized characteristics of tidal channels from a back-barrier New Jersey salt marsh and recommended that these be used to guide initial channel design of Atlantic Coast restoration projects. However, for many areas these data are lacking or only minimally available. In addition, we know little about how creek characteristics vary across the marsh. Data from the bay edge and marsh plain will be essential for designing proper hydrology for a restored wetland.

3.2.4 Freshwater inputs

A significant component of the hydrology of coastal wetlands is the input of freshwater; hence, watershed issues are of critical importance to the restoration and management of coastal wetlands. Under natural conditions in southern California, it is likely that coastal wetlands received freshwater inputs only during winter and early spring, depending on the highly variable annual rainfall. Under present conditions, dry season freshwater inputs (along with sediment) have generally increased due to urbanization and agriculture within the watershed. In some watersheds, however, freshwater inflow has declined. River flows are diverted from northern to southern California, reducing freshwater inflows to San Francisco Bay, and levees along the Mississippi River divert flood flows and sediments away from much of coastal Louisiana. Dams and reservoirs have reduced both total freshwater inflow and the wet-season pulse. Equally important, dams and other activities disturb the sediment budget to the wetland and estuary. It is clear that watershed management must be addressed as part of restoration planning. In Tijuana Estuary, an 8-ha restoration project has been designed in the south arm of the estuary, adjacent to a small watershed that drains directly into this area (Chapter 1, Box 1.11). The estuary's watershed has a history of catastrophic flooding, wastewater spills, erosion, and sediment deposition downstream (Zedler et al. 1992). Restoration plans for the Model Marsh have been integrated with planning for sediment basins and erosion control methods in the watershed in order to avoid direct impacts to this restoration project (Jim King, *personal communication*).

3.2.5 Erosion and sedimentation

Tidal flows and available sediment determine the balance of sedimentation and erosion that occurs across a coastal wetland: within creeks, on mudflats, and in the vegetated areas of the wetland. Sedimentation rates from tidal inputs vary among wetlands and across a wetland, with the highest rates and most dynamic conditions in tidal creeks, and slower rates on mudflats and in vegetated areas. We discuss each of these areas below.

Intertidal mud and sand flats are an integral part of the wetland system. However, research on sediment dynamics and geomorphology of unvegetated mudflats is limited for both natural (Anderson et al. 1981) and restored sites (Simenstad and Thom 1996). Pethick (1996) reviewed hydrology and sediment dynamics for mudflats and concluded that short-term responses to meteorological changes tend to balance out over time, while longer-term shifts due to environmental changes, sea-level rise, or sediment tend to be unidirectional. In a geomorphic framework, Pethick (1996) also noted that the marsh plain acts as a sediment storage zone, supplying or receiving sediment from the mudflat. This relationship is similar to that found in a beach and dune system, although the marsh plain-mudflat system is composed of finer-grained sediments due to the lower overall energy in the marsh compared to beach systems that are driven by large waves.

Intertidal mudflats are flooded more frequently by the tides than vegetated areas, resulting in a longer period for sediments to be deposited. However, mudflats lack plants,

and there is no "baffling" effect to slow water velocities and increase sedimentation rates (Gleason et al. 1979, Frey and Basan 1985). In addition, current speeds over the mudflats can be great, so there is more chance for erosion. Because of these counteracting dynamics, deposition rates on mudflats tend to be highly variable. Accretion rates are assumed to be within the same general range as those found in adjacent vegetated areas; however, few studies have measured either vertical accretion rates or sediment deposition on mudflats or sandflats (Pestrong 1972, Anderson 1973, Frostick and McCave 1979, Anderson et al. 1981). The elevation of the mudflat represents a balance between depositional and erosional forces, resulting from the shallow water wave action across these habitats.

Tidal creeks and channels encompass a wide range of sizes and flow conditions, and sediment dynamics are also highly variable. Within large channels, flow rates can be high, resulting in extreme rates of erosion and sedimentation until a hydrologic balance is established at a restoration site. Smaller creeks will be less variable, although they also can be very dynamic in the early phases of restoration. At the Tidal Linkage Tijuana Estuary, a natural tidal creek was extended by dredging a deeper and broader channel from upland topography (Box 1.10). In one year (the ENSO winter of 1997–98), 30 to 50 cm of fine sediment accumulated in the bottom of the new channel. Even without winter storms, sediment redistribution within the creeks can be rapid if there are high rates of flow and resuspension of unconsolidated, fine sediments. The elevation of the restored creek bottom will eventually match that of the adjacent natural creeks. Overexcavation of creeks by about 10% to allow sediment accumulation is a good way to establish appropriate sediment texture in restored creeks, and this will benefit invertebrate establishment and growth (Chapter 5).

Although natural coastal wetlands tend to be depositional, accretion rates are much slower on the marsh than in tidal creeks, with annual rates typically ranging from <0.1 to 1 cm/yr (Table 3.1; also see Stevenson et al. 1986, Reed 1990, Callaway et al. 1996*b*). Studies of sediment accretion have been completed primarily on the East and Gulf coasts of North America. Sediment deposition occurs both through chronic tidal inputs and storm pulses. Rivers supply sediments over the entire area that is flooded, while sea storms cause dune overwash of sandy sediments into more localized areas, namely the nearby channels. In most systems, accretion rates are greatest adjacent to tidal creeks, with very slow rates of tidal accretion in infrequently flooded parts of the marsh (French and Spencer 1993). Craft et al. (1993) found higher accretion rates adjacent to tidal creeks; however, accretion rates were greater in irregularly flooded areas of the wetland than in regularly flooded areas due to high rates of organic matter accumulation in irregularly flooded portions of the wetland. In most natural systems, accretion rates are similar to increases in relative sea level. Changes in elevation are very gradual, except in areas with (1) extremely high rates of subsidence, e.g., coastal Louisiana and parts of Chesapeake Bay, (2) large storm-related inputs, e.g., southern California (Cahoon et al. 1996), or (3) tectonic activity, e.g., in Alaska (Thilenius 1990), Oregon (Darienzo and Peterson 1990), and Chile (Reed 1989).

3.2.5.1 Storm sediment inputs

Storm inputs of sediment can be a significant component of the overall sediment balance of some coastal wetlands (Stumpf 1983); this is especially true in southern California where watersheds are small and steep-sloped, have been modified, include highly erodible soils, and have high-velocity flows immediately following rainfall events. Rainfall is highly seasonal with high interannual variability (Florsheim et al. 1991, Trimble 1997). Sediment inputs from storm events are frequently orders of magnitude greater than sediment deposition under normal tidal action (Stumpf 1983); however, storm inputs are very unpredictable, both temporally and spatially. Sediment accretion resulting from Hurricane

Table 3.1 Sediment accretion rates from coastal wetlands around the world. Accretion rates were measured using ^{210}Pb (100-year time scale), ^{137}Cs (20- to 35-year time scale), and marker horizons (1- to 10-year time scale).

Location	Author	Method used	Range of sediment accretion rates (cm/yr)
		Atlantic Coast	
Multiple sites, ME	Wood et al. 1989	Marker horizon	0–1.3
Nauset Marsh, MA	Roman et al. 1997	^{210}Pb, ^{137}Cs, Marker	0–2.4
Waquoit Bay, MA	Orson and Howes 1992	^{210}Pb	0.3–0.5
Narragansett Bay, RI	Bricker-Urso et al. 1989	^{210}Pb	0.2–0.6
Barn Island, CT	Orson et al. 1998	^{210}Pb, ^{137}Cs, Marker	0.1–0.4
Farm River, CT	McCaffrey and Thomson 1980	^{210}Pb	0.5
Long Island Sound, CT	Anisfield et al. 1999	^{210}Pb, ^{137}Cs	0.1–0.7
Flax Pond, NY	Armentano and Woodwell 1975	^{210}Pb	0.5–0.6
Great Marsh, DE	Church et al. 1981	^{210}Pb	0.5
Delmarva Peninsula, VA	Kaslter and Wiberg 1996	^{210}Pb	0.1–0.2
Chesapeake Bay, MD	Stevenson et al. 1985	^{210}Pb	0.2–0.4
Chesapeake Bay, MD	Kearney and Ward 1986	^{210}Pb	0.5–0.7
Chesapeake Bay, MD	Kearney and Stevenson 1991	^{210}Pb, ^{137}Cs	0.3–0.8
Chesapeake Bay, MD	Kearney et al. 1994	^{210}Pb, ^{137}Cs	0.3–0.8
Chesapeake Bay, MD	Griffin and Rabenhorst 1989	^{210}Pb	0.4–0.8
Pamlico Sound, NC	Craft et al. 1993	^{137}Cs	0–0.5
Sapelo Island, GA	Letzsch 1983	Marker horizon	0.2–0.7
Everglades, FL	Craft and Richardson 1998	^{210}Pb, ^{137}Cs	0.1–0.8
		Gulf Coast	
Barataria Bay, LA	DeLaune et al. 1978	^{137}Cs	0.8–1.4
Barataria Bay, LA	Hatton et al. 1983	^{137}Cs	0.3–1.4
Barataria Bay; Fourleague Bay, LA	Baumann et al. 1984	Marker horizon	0.6–1.5
Terrebonne Basin, LA	Nyman et al. 1993	^{137}Cs	0.7–1.8
Calcasieu Lake, LA	DeLaune et al. 1983	^{137}Cs	0.7–1.0
Multiple sites, LA	Cahoon and Turner 1989	Marker horizon	0.4–1.3
Multiple sites, LA	Nyman et al. 1990	^{137}Cs	0.6–0.9
Multiple sites, LA	Cahoon 1994	Marker horizon	0.1–1.1
Multiple sites, LA	Bryant and Chabreck 1998	^{137}Cs	0.3–0.9
Multiple sites, FL, MS, TX	Callaway et al. 1997	^{137}Cs	0.2–0.9
Rookery Bay, FL; Terminos Lagoon, Mexico	Lynch et al. 1989	^{210}Pb, ^{137}Cs	0.1–0.5
		Pacific Coast	
Tijuana Estuary	Cahoon et al. 1996	Marker horizon	0.1–8.5
San Francisco Bay, CA	Patrick and DeLaune 1991	^{137}Cs	0.4–4.2
Multiple sites, OR & WA	Thom 1992	^{137}Cs	0.2–0.7
		Europe	
Severn Estuary, England	French et al. 1994	^{210}Pb	0.4
Scolt Head Island, England	Stoddart et al. 1989	Marker horizon	0.1–1.4

Table 3.1 (continued) Sediment accretion rates from coastal wetlands around the world. Accretion rates were measured using ^{210}Pb (100-year time scale), ^{137}Cs (20- to 35-year time scale), and marker horizons (1- to 10-year time scale).

Location	Author	Method used	Range of sediment accretion rates (cm/yr)
Scolt Head Island, England	French and Spencer 1993	Marker horizon	0.1–0.8
Eastern Scheldt, Netherlands	Oenema and DeLaune 1988	^{137}Cs, Marker horizon	0.4–1.6
Island of Sylt, Germany	Kirchner and Ehlers 1998	^{210}Pb, ^{137}Cs	0.6–1.5
Multiple sites, England, Netherlands, Poland	Callaway et al. 1996*a*	^{137}Cs	0.3–1.9

Andrew was up to 11 times the long-term annual accretion rate for estuarine wetlands in Louisiana (Nyman et al. 1995). The intensity of storm inputs varies inversely with their frequency of occurrence, i.e., severe storms are extremely rare. Spatial variability can also be extreme. In Tijuana Estuary, storm inputs are very large in some parts of the estuary (Zedler 1983). Haltiner and Swanson (1987) estimated that 75% of the total sediment input for one year (1980) occurred during seven days. Near the mouth of the estuary, storm overwash of dune sediments can bury marsh vegetation and fill tidal creeks (Fink 1987). Callaway and Zedler (*in preparation*) documented deposits of 10 to 30 cm in the south arm from flooding of small, local drainages during the winter of 1994–95. This sedimentation event buried *Salicornia virginica* in areas adjacent to the tidal creek (Figure 3.6). Cahoon et al. (1996) measured 2.5 to 8 cm of sediment accumulation in areas dominated by *Spartina foliosa* in the north arm of Tijuana Estuary during the winter of 1992–93; however, accretion in the high marsh was only 1 to 2 mm. Similarly, during the 1997-98 ENSO winter storms, sedimentation rates in the vegetated wetland at the Tidal Linkage were only 3 to 5 mm but were greater than 5 cm in a nearby area dominated by *S. foliosa* (PERL unpublished data).

In order to reduce impacts from storm sedimentation, the landscape (watershed) scale should be considered in locating coastal wetland restoration projects (Section 7.6.3). However, given the unpredictable nature of these inputs (as well as their spatial patchiness), in some cases it will be difficult to locate restoration sites where there is minimal risk of excessive sedimentation. It is thus important to consider first the source of sediments within watersheds. If soils upstream are continually disrupted, then anthropogenically caused erosion will deliver large amounts of sediments to coastal wetlands. A watershed-based management plan for erosion control may be needed along with the wetland restoration plan. Stormwater detention ponds within the watershed may help control sediment inputs to coastal wetlands. In some cases, it may not be sufficient to slow the release of sediments upstream, and downstream engineering solutions may be needed to capture sediments.

3.2.6 Restoration methods and sediment dynamics

Sedimentation issues must be considered in developing sustainable restoration projects. Sites should be designed to develop over time in order to minimize maintenance problems (Section 7.6). In the last decade, some restoration sites have been overexcavated with the expectation that initial sedimentation will rapidly raise the elevation (Williams *in press*). Coats et al. (1989) recommend overexcavating creeks so that they will be large enough to maintain tidal flows. Much greater velocities are needed to erode sediment of a particular size than to

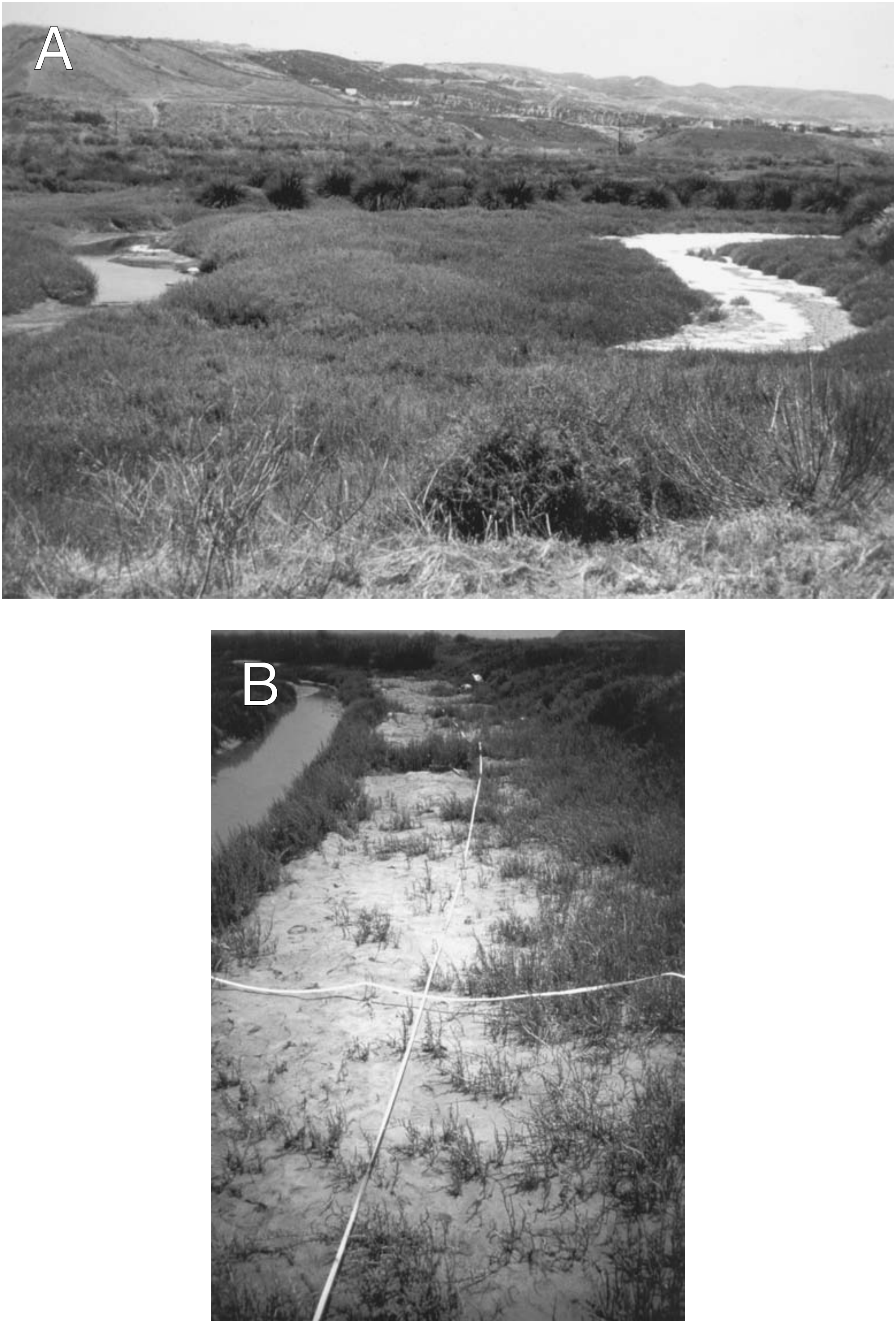

Figure 3.6 Tidal creek and adjacent vegetation of pickleweed (*Salicornia virginica*) in the south arm of Tijuana Estuary before **(A)** and after **(B)** winter storms in 1994–95. The open areas in the foreground of (A) are salt pannes. Large areas of pickleweed along the creek were buried from this sedimentation event.

deposit it, so it is better to create too large a channel than too small a channel. In addition, creek systems are very dynamic and will rapidly build up to an equilibrium condition. If a creek is underexcavated, it will restrict flow and retard the development of a restored site.

A similar approach has been proposed for the marsh plain. This can be achieved either by overexcavating the marsh plain or by breaching levees at subsided sites. In the second case, restoration sites are left at low elevations by design, with the expectation that they will accumulate sediments rapidly (Williams *in press*). Where there is a high load of suspended sediment, this method is very useful, because fine sediments typical of natural marshes are likely to accumulate (Section 3.3.1). Suspended sediment concentrations are very high in San Francisco Bay; marsh plain accretion rates of 2 to 3 cm/yr or more are common until an equilibrium elevation is reached (Jeff Haltiner *personal communication*). However, this method is not recommended for coastal wetlands with small watersheds and/or little sediment input (either naturally or due to damming of natural sediment sources). Under low-sediment conditions, wetlands have very slow rates of accretion. In Say Diego Bay, large areas of the constructed marsh plain eroded rather than accreted sediments (Haltiner et al. 1997, PERL unpublished data, Box 1.8). If overexcavation is considered, early site visits should assess the concentrations of suspended sediment, flow rates, and rates of sediment accretion across different elevations at a nearby natural marsh.

It is not yet possible to predict exact accretion rates for a restoration site. However, Krone's (1987) model of sediment dynamics can be used to predict rates of accretion based on suspended sediment concentrations and frequency of inundation (based on tidal regime and relative elevation). This model was developed to evaluate the historical location of the mean high water datum by simulating changes in wetland elevation over time, and it has been updated to evaluate mineral sediment deposition more generally. Allen (1990, 1995) and French (1993) modified this modeling approach to simulate impacts of sea-level rise on coastal wetlands, and these modifications could be useful for evaluating sediment dynamics of restored wetlands. As data become available from multiple natural and restored sites, a more predictive geomorphologic model of sediment accretion rates could be developed. In addition, restoration planning would be greatly enhanced by having models that integrate both mineral-based processes and organic sediment processes, i.e., compaction, organic matter production, and decomposition (Morris and Bowden 1986, Callaway et al. 1996*b*, Rybczyk et al. 1998). Until the dynamics of erosion and accretion are understood, we cannot recommend overexcavation of sites in areas with low levels of suspended sediment and low rates of accretion. Wetland creation sites should be graded to an elevation close to that found in nearby natural wetlands if the marsh needs to be vegetated quickly. Overexcavation would create subtidal ponds or intertidal mudflats.

An added concern at restored wetlands is sea-level rise associated with global climate change. Designs for restored wetlands should consider the long-term future of the site, preferably by including broad buffer areas with gradual slopes (Box 2.3) to allow for the landward migration of the wetland under increased rates of sea-level rise. The more costly alternative is to plan for water-control structures to maintain necessary water levels.

3.2.6.1 Special case of subsided wetlands

Restoration of full tidal action may not always be advisable (Coats et al. 1989). In areas with excessive subsidence, opening the area to full tidal action would produce subtidal basins and intertidal mudflats, rather than vegetated marsh. Also, some management goals, such as the creation of waterbird feeding areas, require incomplete drainage. Such cases require water control structures, such as gates, weirs, levees, or pumps, to maintain the proper hydrology. However, these structures may lead to hydrologic isolation of the site and actually slow development of natural wetland functions (see Boesch et al. 1994 for a review of issues related to managed wetlands in Louisiana).

Alternatively, sediments (typically from dredging operations) can be used to raise the elevation of the restoration site. This has been done at several created wetlands, e.g., Sonoma Baylands in San Francisco Bay, California, dredge spoil wetlands in Texas (Lindau and Hossner 1981), North Carolina (Broome et al. 1988, Craft et al. 1988), and elsewhere. In using dredge spoils, care should be taken to ensure that sediment texture is correct. In addition, the following factors need additional evaluation: the consolidation of material following drainage (Section 7.6.1), pollutant release, and the stabilization of dredge material. In San Francisco Bay, the preferred approach is to use dredge spoils below the marsh plain, allowing for the upper 20 to 30 cm of material to accumulate naturally. This allows a more natural creek system to develop. In areas where dredge spoils are potentially contaminated, it has been proposed that such material be used below the plant rooting zone and that it be capped with clean sediment.

3.3 Substrate

Although it is well known that substrate conditions directly affect plant growth, invertebrate colonization, and other factors, many restoration projects are constructed using coarse-textured dredge spoil or upland substrates, which are very different from natural marsh soils and sediments. Inappropriate substrates cause numerous secondary problems, discussed below. From several examples, we know that substrate conditions are critical to plant and animal colonization and growth. An analysis of existing soil conditions is needed before restoration begins (Section 3.4.2).

Soils of former wetlands will more likely be suitable for native species than soils at newly created sites. Created wetlands typically have a series of soil deficiencies, primarily related to coarse soil texture, low organic matter content, and occasionally high acidity (Langis et al. 1991, Craft et al. 1991).

3.3.1 Texture

Over the last two decades, it has become clear that many of the substrate-related problems at coastal wetland restoration projects are due primarily to the coarse texture of substrate at created wetlands (Lindau and Hossner 1981, Langis et al. 1991). Because natural wetlands develop in areas with slow-moving tidal water, their soils have a significant clay component (Pethick 1984). Coarser material tends to accumulate adjacent to creeks, with a gradual increase in finer particles across the marsh; however, even at the edge of the marsh where sandier soils are found, there are still significant amounts of clay and silt.

Created wetlands tend to be located on sandy, coarse substrates, usually either dredge spoil material or soil from excavated uplands. Lindau and Hossner (1981) found that soils from dredge spoil wetlands in Galveston Bay, Texas, had significantly less clay at middle and high elevations compared to three natural reference wetlands. At the lowest elevations, textures were similar, due to the naturally high sand content in these soils. Because of differences in drainage, water holding capacity, nutrient retention, cation exchange capacity, and other factors, coarse substrate frequently leads to many additional substrate problems at created wetlands, as described below (Figure 3.7).

3.3.2 Organic content

One of the major compounding problems of coarse soils is that they do not accumulate organic matter rapidly. Compared to conditions in natural wetland soils, coarse soils are well drained and relatively aerobic. Under aerobic conditions, decomposition rates are

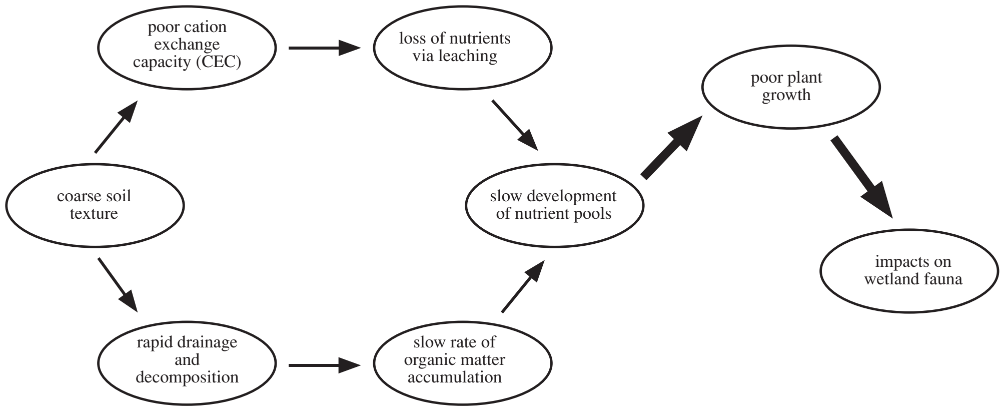

Figure 3.7 Flow chart of the likely consequences of incorrect substrate texture at a restored wetland.

high and organic matter does not accumulate within the soil, even if organic matter production rates are similar (Hemminga et al. 1988).

Soil organic matter plays a key role in the structure and chemistry of wetland soils. Most natural salt marsh soils have from 10 to 40% organic matter. Organic matter content of southern California wetland soils is lower, from 5 to 10% (Langis et al. 1991), perhaps due to year-round decomposition and, in some places, large inputs of mineral sediments. Soil organic matter is essential to nutrient cycling because it provides the main pool of soil nutrients, with continual availability via slow mineralization of organic matter (Follet et al. 1981, Brady and Weil 1999). Organic matter also improves structure for wetland soils by contributing to the aggregation of fine particles, adding to the soil stability, and decreasing soil bulk density (Tate 1987). Chemically, organic matter increases cation exchange capacity, which determines availability of cations, such as NH_4^+, and buffers soil pH. Finally, organic matter supports soil microbial activity, including nitrogen fixation (Langis et al. 1991).

3.3.3 Nutrients

The limiting nutrient for most coastal wetlands is nitrogen because nitrogen is readily lost through denitrification and because phosphorus is usually plentiful (Valiela and Teal 1974, Mitsch and Gosselink 1993). Because soil organic matter is often low at restoration sites, nutrient concentrations are also low. In natural wetlands, nutrients are recycled from organic pools, via high rates of nutrient mineralization and uptake by plants or decomposers (White and Howes 1994). In restored wetlands, both supply and uptake rates are severely reduced.

Furthermore, when nutrients are available at restored wetlands, they are quickly leached from coarse-textured, inorganic soil, due to low cation exchange capacity. Even after adding organic matter to restored soils, Gibson et al. (1994) found little impact on plant growth because most of the organic matter was mineralized and nutrients were quickly leached. The effects of cation exchange capacity on leaching rates are particularly important for the cation NH_4^+, the inorganic nitrogen component which tends to accumulate in anaerobic wetland soils (Reddy and Patrick 1984). Nitrate is easily reduced to NH_4^+ or lost from wetland soils via leaching and denitrification; thus nitrate rarely reaches high concentrations in salt marshes (Reddy and Patrick 1984, Langis et al. 1991).

3.3.4 Compaction

In addition to coarse texture, restored sites also may have compacted soils. Heavy machinery used to excavate and shape the topography also compacts the sediments (either locally or throughout the site). Soil compaction slows plant growth by impairing root penetration. In addition, water can pond in low spots underlain by compacted soil. This local phenomenon was apparent at a restoration site in Mission Bay (Box 1.5), where standing water in tractor tracks prevented seedling establishment. While topographic heterogeneity is desirable (Chapter 2), unnatural patterns, such as "stripes," detract from the esthetics.

Compaction is most likely to occur when wet substrate is cultivated or graded. Grading the site when it is dry will minimize the loss of soil structure (Brady and Weil 1999). Sites with fine textured soils are also more compactable than sandy soils.

Compacted layers below the surface can hinder plant establishment. At the Tidal Linkage (Box 1.10), a naturally formed hardpan (cemented layer) was encountered when the upland was graded to elevations suitable for the marsh plain. The planting contractor had to use a power auger to break through this layer to plant small sections of salvaged marsh sod. For our field experiment, a tractor was employed to "rip" the substrate and break it into smaller pieces prior to rototilling and planting seedlings.

3.3.5 Soil salinity

The problems of coarse or compacted soils can be compounded by the development of high soil salinities (Bertness et al. 1992, Callaway and Sabraw 1994). Once tidal water is regularly supplied to the restoration site, evaporation will cause salt to accumulate on the soil surface; this is specifically a problem at higher elevations, where soils are exposed for long periods between high tides. High salinities prevent germination and plant establishment, sustaining bare areas in the high marsh. Much of the area intended to support high marsh in San Diego Bay (Box 1.7, Box 1.8) remained unvegetated, with salinities exceeding 100 ppt. Soil amendments (decomposed kelp) and irrigation were necessary prerequisites for plant establishment.

3.3.6 Soil pH

Natural coastal wetlands tend to have soils that are well buffered and neutral or slightly acidic (Ponnamperuma 1972). Restoration sites may have widely ranging pH because of oxidation prior to restoration (e.g., exposed dredged spoil material or drained wetland soils). In some cases, acid sulfate soils or "cat clays" are a major concern for coastal wetland restoration. Low pH develops where sulfides accumulate and then, upon exposure from dredging or drainage, become oxidized to sulfuric acid. Tidal flooding does not always correct the problem, and liming of soils may be necessary to raise soil pH. Our ability to recommend treatment of pH problems is limited, because experimentation is rare. Freshwater wetlands that have become acidified are not necessarily treatable (van Duren et al. 1998).

3.4 Planning considerations

3.4.1 Hydrologic assessments

The existing hydrology of potential restoration sites needs to be analyzed to determine which alterations are suitable (Williams 1986, Coats et al. 1995). Potential constraints on restoration are existing creek depth, creek density, sediment conditions, and inputs of freshwater or sediment. Questions to consider include:

- What is the relationship of the current site topography to that of the tidal range?
- How much excavation or placement of fill will be needed?
- Are the existing creeks large enough to handle additional flows or will sufficient scouring take place to accommodate increased tidal exchange?
- Are there bottlenecks or sills that will limit flow to the restored site?
- How will increases in tidal prism affect the existing network of tidal creeks in adjacent natural areas?
- Will the restoration be affected by excessive sedimentation or will there be suitable suspended sediment concentrations to promote high rates of sediment accumulation in areas that are planned for overexcavation?
- What are relative inputs of freshwater and seawater at the site?

Aerial photographs, topographic maps, and other remote data are useful, but field visits are essential to confirm preliminary assessments (see Section 2.2 and 2.3 for a discussion of historical records and current site conditions). In addition, hydrologic modeling of projected conditions at the site can be useful in identifying potential problems (Coats et al. 1989).

3.4.2 *Identification of proper substrates*

Considering the importance of substrate conditions, better information from natural reference marshes would improve wetland restoration efforts. Substrate conditions (texture, organic matter content, nutrient concentrations, salinity, etc.) should be analyzed and included as part of this reference information (Chapters 2, 6). We need much better basic information for how these parameters vary across natural marshes, from tidal creeks inland, and how variability in soil parameters affects plant growth. Basic information from natural sites would be very useful in designing restoration sites and for evaluation of plant responses so that problems could be avoided at future restoration projects.

3.4.3 *Evaluating existing conditions at restoration sites*

Several qualities of the substrate at the proposed restoration site need to be known before extensive restoration plans are developed. To date, substrate testing has focused on texture to determine options for disposing of excavated material. In southern California, sand can be disposed on beaches, while silt and clay must be trucked off site. Little thought has been given to the substrate that is left to support ecosystem restoration. Future testing should target the substrate at the depth projected for marsh and creek surfaces. In the case of dredge spoil projects, the spoils that will be used for "marsh building" should be tested for texture, organic matter content, nutrient concentrations, contaminants, and hypersalinity (low salinity is not a problem). If the spoils are not appropriate for wetland development, alternative sites or sources of soil should be considered, or amendments proposed. Because soils are very slow to develop, substrate is frequently the limiting step for marsh restoration.

An option at many sites is to import fine material ("top soil") to the restoration project (Brown and Bedford 1997). At mitigation projects, a potential source of fine soil is the marsh that is permitted for development. Use of salvaged marsh soil would also incorporate rhizomes, roots, seeds, and microorganisms into the restoration site, thereby enhancing plant establishment and growth (Chapter 4). In projects where marsh is created from upland, any fine soil that is excavated on site can be salvaged and replaced at the finished elevation. The site should be overexcavated to accommodate backfilling of this fine material. At Tijuana Estuary, the Tidal Linkage experimental site (Box 1.10) had very

coarse substrate. We improved overall soil texture by overexcavating the marsh plain slightly and adding fine sediment from a channel that was dredged through the adjacent mudflat. We then used a rototiller to mix the fine and coarse sediments prior to planting. A small experiment tested the effect of adding fine material and showed that plant cover was significantly increased after 2 years. Although stockpiling of fine sediments increases the amount of excavation and handling of sediments, it reduces the cost of disposing of fine sediments. In the long term, the ecological cost is less because proper soils accelerate ecosystem development.

3.5 Restoration solutions

3.5.1 Hydrology

It is extremely difficult to make hydrologic modifications once a site is under tidal influence; therefore, adequate hydrologic conditions should be in place when tidal action is initiated. Creeks and channels will evolve and develop over time, so it is not possible to design a static system. However, there must be adequate tidal flow for these developments to take place (Williams 1986, Williams *in press*). As indicated above (Sections 3.2 and 3.4.1), considerations should be made for sufficient tidal prism, adequate creek density, and creek morphology. In addition, landscape issues should be considered to ensure that watershed inputs (freshwater, sediments, and nutrients) are suitable for long-term sustainability of the restored wetland.

In order to develop the best hydrologic design, the planning and design process should carefully consider issues related to the hydrology and geomorphologic development of the site. We recommend the following steps, in order:

- Consider the tidal range that is available at the site.
- Evaluate other hydrologic inputs (natural freshwater inputs, storm water, etc.) that will affect how the restored wetland will develop.
- Consider any constraints on making hydrologic modifications (Section 2.3.2).
- Determine the range of elevations for the marsh plain in nearby natural wetlands.
- Establish a design elevation for the marsh plain based on the above steps. This is the desired elevation of the marsh plain after it has reached equilibrium.
- Select the construction elevation for the marsh plain, based on the design elevation, the estimated rate of initial sediment accretion, and the time period allowable for wetland development.
- Design the tidal creeks and channels within the restored wetland, considering the issues outlined in Section 3.2.3.

In designing the tidal creeks and channels, the first consideration should be the order and pattern of creeks and channels. Once creek order and pattern have been determined, the proper channel sizes should be determined, based on the desired tidal flow at the site (see Williams 1986, Coats et al. 1995 for more detail on these considerations). This process should be based primarily on the hydrologic conditions found at nearby natural wetlands.

3.5.2 Soil amendments

The first priority is to obtain soil that is similar to that of the reference wetland; where that is not possible, amendments should be considered. Gibson et al. (1994) found that a one-time addition of organic matter had a very short-lived effect on *Spartina foliosa* growth at Marisma de Nación, San Diego Bay. Material that was added to the soil decomposed

within 5 months. Boyer and Zedler (1998) found that biweekly nitrogen additions (urea fertilizer) stimulated growth immediately, but little of the nitrogen was retained into the second year. More recent data indicate that even 5 years of nitrogen addition do not continue to enhance plant growth once amendments cease (PERL, unpublished data).

There are still many gaps in our understanding and use of soil amendments. Additional experiments are needed to evaluate the use of different types of amendments before they can be recommended on a general basis. Priority should be on evaluating amendments that are cheap and widely available since their use across large restoration sites will involve large quantities. For wetland creation projects that involve the destruction of natural wetlands, salvaging and stockpiling of natural marsh soil for future use should also be evaluated further (Sections 3.4.3 and 4.3.4). Because of the improved growing conditions with fine, organic rich soils (as well as the potential for establishment through seeds and rhizomes), the use of salvaged soils could substantially improve the development of created wetlands

3.5.2.1 Texture amendments (clay)

Given the problems with coarse soil at wetland restoration sites, clay and silt are obvious potential amendments. Because fine-texture amendments are not readily available, stockpiling of excavated material is a potential source, as discussed above. A second source of fine substrate is fine-textured dredge spoils.

However, toxicity is a potential problem for fine-soil amendments, especially dredge spoils from industrial harbors. Because fine soils are chemically reactive, they tend to accumulate pollutants (including both heavy metals and organic pollutants) much more rapidly than coarse soils. Before fine-texture amendments are used at a site, they should be tested for potential toxicity to both plants and animals. This is a special concern if sediments are excavated from subtidal, anaerobic environments and are exposed to aerobic conditions because this will rapidly increase mobility of metals and other potential pollutants (Gambrell and Patrick 1978). In any case, fine texture amendments require close monitoring if they are stockpiled. Material that becomes oxidized will lose organic matter and nutrients. Furthermore, if clays are allowed to dry completely, they will become extremely hard and unsuitable for use in restoration.

3.5.2.2 Organic matter amendments

We have used composted kelp to increase rates of plant establishment and growth for two restoration projects in San Diego County. At San Diego Bay (Box 1.7), kelp was used in combination with freshwater irrigation to establish high marsh plants in areas with highly compacted and saline soils. Kelp was worked into the soil to a depth of approximately 30 cm, and the increased organic matter reduced soil bulk density, improved water retention, and increased soil structure when it was incorporated into the soil.

Composted kelp was also useful in sandy soils at the Tidal Linkage (Box 1.10). In plots planted with eight species of the marsh plain, plant cover, height, and layering were significantly greater in kelp-amended plots (see Figure 4.14). Some of these nutrients will be leached out of the sandy soils; however, amended plots remained more luxuriant after two growing seasons. Composted kelp is readily available in southern California, and we recommend its use in this region. In areas where kelp may not be readily available, other organic amendments that improve soil structure, organic content, and nutrient status should also be evaluated.

Sewage sludge is another possible soil amendment. It is high in nutrients and readily available; however, some sources can have high concentrations of heavy metals (e.g., cadmium) or other contaminants. Sewage sludge has been used in long-term fertilization

experiments (Valiela et al. 1975, 1976), with large increases in biomass after multiple years of additions; however, effects from single amendments at restoration sites have not been evaluated.

Alfalfa and straw have both been used to increase organic matter content of wetland soils (Gibson et al. 1994). Straw has low nitrogen content and microbes can colonize the organic matter and further immobilize nitrogen. Amendments that are low in nitrogen also have slow decomposition rates, with little release of nutrients over time (Brady and Weil 1999). Sawdust or shredded bark are possible organic matter amendments, but these materials have even lower nitrogen content. Because of these properties, these types of amendments would not be useful in most wetland restoration projects where nitrogen is usually limiting. However, in special cases where there is excessive soil nitrogen (old sewage ponds, areas with excessive nitrogen inputs), sawdust and bark may improve soil properties while reducing excessive nutrient effects.

3.5.2.3 Nutrient amendments

In wetland soils, the focus of most fertilizer additions has been on nitrogen rather than phosphorus, because nitrogen is consistently limiting to plant growth in salt marshes (Valiela and Teal 1974, Mitsch and Gosselink 1993). In southern California, Covin and Zedler (1988) found that nitrogen additions increased growth of both *Spartina foliosa* and *Salicornia virginica* when grown alone, although *S. virginica* was the superior competitor for nitrogen when grown together. At the Connector Marsh in San Diego Bay (Box 1.7), Boyer and Zedler (1998) found short-term effects of nitrogen additions on *S. foliosa* but little long-term impact on aboveground growth. In addition, there were shifts in community dynamics with fertilization; added nitrogen promoted growth of the annual *Salicornia bigelovii* over *S. foliosa* (Boyer and Zedler 1999; Figure 3.8). It does not appear that fertilizer promoted significant accumulation of either belowground biomass (Boyer et al. *in review*), soil organic matter, or nutrients in sandy dredge soils. Given these shortcomings, the long-term ability of fertilization to create a self-sustaining system is questionable, especially without proper soil texture. Fertilization is recommended only in special cases, where it is likely that nutrients will be retained over the long term and where it will not cause shifts in community composition.

3.6 Summary

Establishing proper hydrology is essential for restoring a functional wetland. The initiation of tidal flows will reinstate the geomorphic processes that drive coastal wetland dynamics; however, careful consideration of the fine points of wetland hydrology is necessary in order to maximize the progress of a restored site. Early restoration projects did not incorporate the fine-scale heterogeneity of natural coastal wetlands, and future efforts to restore coastal wetlands should include more complex hydrologic systems. Elevations at a restored wetland should be established so that the site will develop as quickly as possible, in a manner that mimics the natural development of coastal wetlands. The extent of overexcavation and natural development will depend on sedimentation rates at the site, as well as the temporal expectations for restored wetland functioning. In order to enhance the sustainability of the restored wetland, the relationship of the wetland to the watershed also must be considered. A series of steps for the hydrologic design of coastal wetland restoration projects is outlined in Section 3.5.1.

Soils are the key to plant establishment and growth; however, they have been overlooked at many restored and created wetlands. Restored sites with natural wetland soils are most appropriate for native plant growth, while created wetlands represent the greatest

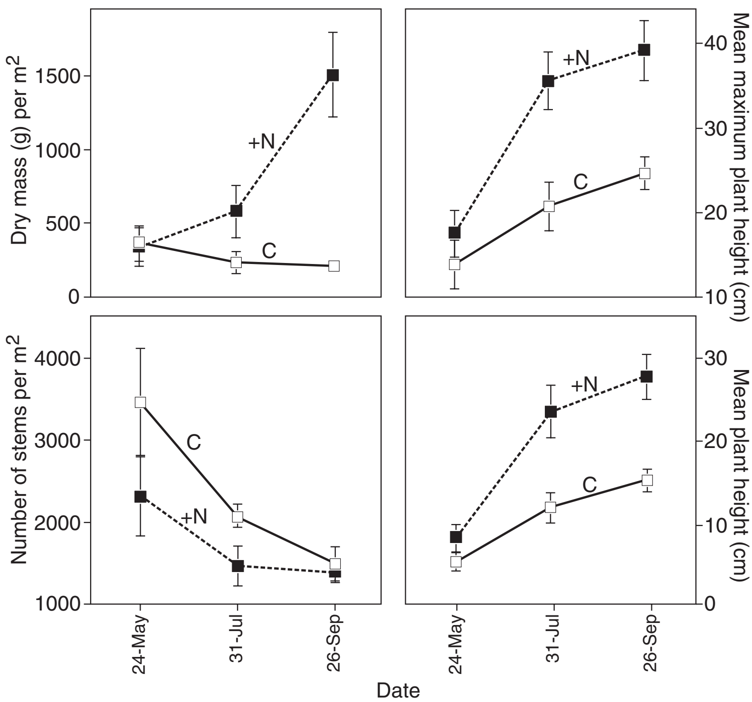

Figure 3.8 Aboveground biomass, stem density, maximum height, and mean height for *Salicornia bigelovii* growing in plots with *Spartina foliosa*, with nitrogen fertilizer added (+N) and without fertilizer (C). *Spartina foliosa* responded to N additions only when *S. bigelovii* was absent. (From K. E. Boyer and J. B. Zedler, Nitrogen addition could shift plant community composition in a restored California salt marsh, *Restoration Ecology* **7:**74-85 (1999). Reprinted by permission of Blackwell Science, Inc.)

challenge for establishing proper soil conditions. In creating and restoring coastal wetlands, we should evaluate both the physical (texture, compaction) as well as the chemical and biological parameters of the soil (salinity, organic content, nutrient concentrations, and pH). Soil texture affects many soil processes, and special attention should be given to it because texture will not change over time, except through accumulation of new sediment. With improper soil texture, many problems are likely to develop that can severely limit wetland functions (Figure 3.7). Where possible, wetland soils should be salvaged and stockpiled from impacted sites so that this material can be used in the establishment of restored or created wetlands. Where salvage is not possible, amendments should be made, or natural sediments allowed to accumulate (if the time frame of the project is appropriate). Hydrology and soils are linked through geomorphologic development; thus, these factors should be considered simultaneously and the respective plans carefully coordinated.

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chapter four

Establishing vegetation in restored and created coastal wetlands

Gary Sullivan

4.1 Introduction

Along the Pacific Coast of North America, the vast majority of marshland has been developed or degraded over the past 100 years (Dahl 1990). The remaining fragmented coastal wetlands, still pressured by developers, constitute a rare and invaluable resource. The coastal wetlands of southern California are increasingly disturbed, reduced, fragmented, and isolated, with plant diversity being lost in the process. By way of example, we present data from ten of San Diego County's wetland systems that were censused to compare lists of indicator plant species in 1998 with historical records summarized in 1990 (Table 4.1). Six of the ten sites had fewer species in the recensus, with losses reaching 36%. Data such as these indicate the urgent need for restoration and expansion of wetland habitat. Plants in existing marshes should be protected, and where populations cannot be sustained, the plants should be salvaged and used to increase the long-term prospects of marsh restorations.

Restoring a coastal wetland ecosystem might include excavating fill from a degraded wetland, reintroducing species that were once present, breaching dikes, opening tide gates to resume tidal inundation, or some combination of these approaches. Once wetland hydrology and substrate are restored, the next step is to establish the appropriate vegetation. The plant community directly or indirectly performs many of the biologically and economically desirable functions of wetland ecosystems (Mitsch and Gosselink 1992, Gopal and Mitsch 1995). Wetland plants also serve as the matrix in which the microbial and animal community is embedded. Consequently, a self-sustaining plant community is a primary goal, and vegetation establishment is the most common performance standard. In this chapter, we discuss issues and recommendations in

- developing a planting strategy, including the design approach, choice of species, and source of plant material;
- acquiring and propagating plants, salvaging sods and wetland soils;
- introducing plants to the site, including planting times, preparations for the salt marsh environment, and techniques for outplanting; and
- maintaining the genetic diversity of local populations.

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Table 4.1 The presence of salt marsh indicator species in ten San Diego County coastal wetlands. Historical occurrences were summarized by PERL (1990); the 1998 census was done by Sullivan and Noe (Appendix 3). Key to symbols: X = Recorded prior to 1990 and still present in 1998; x = not recorded prior to 1998; O = no longer present in 1998, I = introduced prior to 1990; and i = introduced after 1990. Wetland abbreviations are given in Appendix 2.

	Coastal Wetland									
Species	TJE	SWM	KFR	LPL	SDL	SEL	BL	AHL	SLR	SMR
Atriplex watsonii	X	X	x	X						O
Batis maritima	X	X	X							
Cordylanthus maritimus	X	X								
Cressa truxillensis	X	X		X	X	X	x	X		X
Cuscuta salina	X	X	X	X	x	X	X	X		O
Distichlis spicata	X	X	X	X	X	X	X	X	X	X
Frankenia salina	X	X	X	X	X	X	X	X	O	X
Frankenia palmeri	I	X	i							
Jaumea carnosa	X	X	X	X	X	X	X	X	X	X
Juncus acutus	X	X		X	X	X	X	O	X	X
Lasthenia glabrata	X	X		X	O	X	O			O
Limonium californicum	X	X	X	X	X	X	O			X
Monanthochloe littoralis	X	X	X	X	X	X	O	O		X
Salicornia bigelovii	X	X	X							O
Salicornia virginica	X	X	X	X	X	X	X	X	X	X
Salicornia subterminalis	X	X	X	X	X	X	X	X		X
Spartina foliosa	X	X	X		i		I			
Suaeda esteroa	X	X	X	O	O			O		O
Triglochin concinna	X	X	X							
Fraction Remaining	18/18	19/19	13/13	12/13	9/11	11/11	7/10	7/10	4/5	9/14
Percent Lost	0	0	0	8	18	0	30	30	20	36

4.2 *Developing a vegetation strategy*

Planting strategies may need to differ among regions and vegetation types (trees vs. nonwoody plants), as well as for different project goals (e.g., mandated mitigation or restoration for other purposes) and different performance criteria (establishing plants, providing specific ecosystem functions, etc.). Ideally, the plant community will develop the structure and function of reference marshes of similar size and type. A chief constraint is inadequate funds to purchase, collect, or propagate the appropriate plant material, to implement the planting scheme, and to establish monitoring and adaptive management procedures. Other constraints include the availability and size of the site, the period of time allotted to accomplish goals and meet monitoring requirements, the availability of plant material for establishing the desired vegetation, and the proximity of the site to a source of propagules for natural establishment.

We strongly recommend that a monitoring program be implemented within an adaptive management plan, in order to assess progress and correct problems as they arise. Many wetland restoration efforts fail to achieve their objectives due to unpredictable events in stochastic environments, and most sites require some management or intervention (Lewis 1982, Mitsch and Wilson 1996).

4.2.1 *Planting strategies*

While specific strategies may differ, there are two general approaches for establishing vegetation (see Odum 1989, Mitsch and Cronk 1992, Mitsch and Wilson 1996, Palmer et al. 1997). Designed wetlands specify plantings in specific areas in relative densities similar to the desired endpoint — as in landscape architecture. The expectation is that vegetation (and function) will develop as desired within the monitoring time frame, a critical goal in mitigation programs. Alternatively, self-designed systems rely more on the species finding suitable habitats following either natural recruitment or the introduction of many species. Examples include created marshes that are left unplanted, or areas where tidal flushing is reestablished to diked areas, allowing existing vegetation to respond (Sinicrope et al. 1990, Frenkel and Morlan 1991). Self-designed systems are allowed to develop via invasion and establishment processes, as dictated by the local topographic, hydrologic, and geochemical environment. Ideally, natural processes would select the species best suited to individual microhabitats.

We recommend combining aspects of designed and self-designed wetlands by maximizing diversity and creating heterogeneous topography, allowing natural processes to determine the pace and direction of community development. Species should be matched to microhabitats when possible, so that assemblages will include species that naturally interact together (Section 4.4.3.2). Community development may be enhanced by the dispersal and establishment of native species from external sources or remnant populations within the restored system.

Because most biotic and abiotic wetland characteristics take a long time to develop (Scott 1995, Gilbert and Anderson 1998), it is often desirable to skip the early stages of ecosystem development and create a mature stage at the start of the project. Adequate soil organic matter levels may not develop for 40 years or more (Craft et al. 1988, Zedler and Callaway 1999), but soil amendments might be incorporated during construction to promote plant growth and increase soil organic content (Chapter 3). Similarly, assembling species in appropriate habitats should facilitate the development of functional communities. Species performance can vary greatly with even small changes in the microhabitat environment (Leigh 1982, Silander 1985, Pennings and Callaway 1992, Vivian-Smith 1997). When plantings take place without regard to habitat preferences, less adapted species will perform poorly or will not persist, especially in the face of competition with species better adapted to the site. Bare areas will be subject to invasion by the weedier native and exotic species, or open patches may simply remain unvegetated as soil salinity increases and promotes the development of a salt panne (Bertness 1991, Bertness et al. 1992, Noe 1999). Multiple plantings should be planned to counter the mortality that occurs in all projects. For example, the overall survivorship of seedlings planted in April 1997 at the Tidal Linkage was greater than 95% after 4 months, but had dropped as low as 36% within individual 2×2 m plots (Figure 4.1). Subsequent replanting increased densities to target levels by the end of the year.

4.2.2 *Species choice*

4.2.2.1 *Maximizing diversity*

Planners must decide which species to include in a restoration project. The overwhelming tendency has been to introduce uniform species mixtures into homogenous landscapes (Gilbert and Anderson 1998), with the number of species chosen much smaller than found in natural systems. Recent restoration projects at San Diego Bay and Los Penãsquitos Lagoon

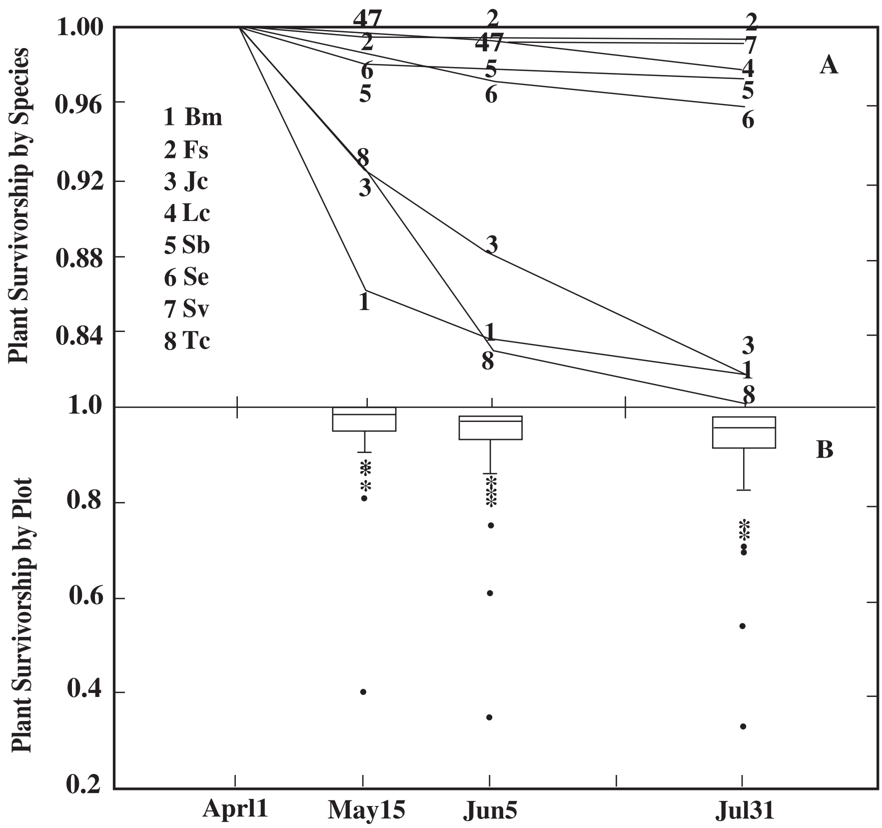

Figure 4.1 Species survivorship at the Tidal Linkage after 4 months, (A) averaged across all plots, and (B) averaged across species within plots. Survivorship stabilized after July 31 until algae and coots damaged the site in December 1997 (Chapter 7). Species abbreviations are in Table 4.4.

typically included 4 to 5 species, rather than the 8 to 12 commonly found on the marsh plain (Appendix 3). Plantings in high marsh and transitional habitats have been even less diverse or entirely ignored. Although this approach simplifies design, it will not likely produce a wetland that functions like the reference system within the planned time frame.

Natural marshes are rich in species, and plantings should include natural levels of diversity, rather than expect all species to establish on their own. Where a few commonly occurring species are introduced, there is strong potential for other desirable species to be competitively excluded. It is often assumed that using the most visible and easily established species is the quickest and most economical way to meet performance requirements, and that including species representative of major functional groups will be sufficient to establish system functions (Walker 1992, 1995, Körner 1993, Cowling et al. 1994). However, recent research shows that apparently similar species may possess unique character profiles and/or function within different areas of the spatial-temporal landscape (Gitay et al. 1996, Sullivan and Zedler 1999). Just as populations with insufficient genetic diversity may be unable to adapt to changing conditions, ecosystems with insufficient species diversity may lack the ability to sustain function in dynamic environments (Tilman et al. 1997). Building diversity into restoration plans biases the ecosystem toward long-term resiliency and sustainability.

One way to foster diversity is to avoid planting a local dominant, if it is likely to invade on its own. This approach might delay its arrival and spread (Section 4.4.3.1). If

planting then emphasizes rarer or dispersal-limited species, a diverse assemblage should be able to establish before the more competitive species dominate the site. This approach is being used in planting the Model Marsh at Tijuana Estuary (Box 1.11). *Salicornia virginica* is being allowed to establish on its own, because propagules are readily available from the surrounding monotype and because we know that it readily colonizes restoration sites and outcompetes other native plants.

4.2.2.2 Using reference sites

Reference sites should be used to develop the blueprint for planting. The reference site will suggest the list of species to consider, their relative abundances, the make-up of assemblages, and the distribution of species across different microhabitats. Reference systems also suggest the diversity of microhabitats that should be included, including their relative distribution and aerial coverage, the types of habitats that are found adjacent to one another, and the physical-chemical properties of each microhabitat (Chapter 2). See NRC (1995) for discussion of these properties.

In reference coastal wetlands, a wide range of conditions may be found between low water and upland habitats. Relatively small differences in slope, topography, and distance from shore or creek edge interact with the physical-chemical environment to create a mosaic of microhabitats (Zedler et al. 1999). Water drains more quickly from steeper slopes, and plants located farther from the water's edge are submersed for less time. Porous sandy sediments drain more quickly, retain fewer nutrients, and have higher redox potential. Sediments remaining saturated for longer periods on flats or depressions may be higher in organic content and relatively anaerobic. Some substrates may be entirely organic (Shaffer et al. 1992, Sasser et al. 1996). Wetland plants have adapted to this varied landscape, and individual species may prefer microsites with different elevation, water level, and/or soil saturation characteristics (Silander and Antonovics 1982, Russell et al. 1985, Pacala and Tilman 1994, Vivian-Smith 1997). That native halophytes have different median or modal elevations indicates their preference for different environmental conditions (Appendix 4). Moreover, the modal elevations of species in reference systems suggest the relative position for planting species in restoration sites.

4.2.3 Plant stock

4.2.3.1 Nursery stock vs. local material

Plants introduced to restoration sites commonly come from nurseries in 2-inch to 1-gallon containers (0.1 to 4.0 liter), and plants are typically placed in a regular matrix on 3 to 4 foot centers (1.0 to 1.3 m), with higher densities occasionally used to establish cover more quickly. Plants from nurseries may establish well initially, but there are some long-term disadvantages to their use. Nursery stock is generally of unknown origin and may come from areas very different from the restoration site (McMillan 1975, Gilbert and Anderson 1998), even from different areas of the country (e.g., plants grown from seed ordered through catalogs). Plants originating in different areas will not be well adapted to local conditions. Plants grown from the seed or cuttings of second- or third-generation nursery stock may be better adapted to greenhouse conditions than to a natural marsh (Lewis 1990). Asexually produced plants may also be undesirable due to their inherently low genetic diversity (Section 4.5.1).

A potentially cost-effective and ecologically sound alternative to purchasing plants of unknown origin is to propagate plants from local stocks (as done for the Tidal Linkage, Box 1.10). This can be done by salvaging existing material in the area to be developed or collecting seed and other plant material from nearby areas, for germination or rooting (Zedler 1996). Where material is available, many nurseries will contract to grow plants

from local seed or specialize in providing local wetland stock (Gilbert and Anderson 1998). Effective propagation techniques will differ among species based on their individual characteristics. For example, some species do not establish well from stem cuttings (e.g., *Salicornia subterminalis*) and others are not easily grown from seed (e.g., *Batis maritima*). However, almost all species establish readily from plugs of sod.

Use of local material may enhance the long-term sustainability and diversity of planted vegetation. Plugs or cuttings should be collected at the site prior to construction (when possible) or from nearby local populations large enough to absorb the loss without endangering that community. Planting local stock (plugs, cuttings, or seedlings grown from locally collected seed) ensures the introduction of locally adapted genotypes and avoids the breakup of coadapted traits. One distinct advantage of using plants grown from seed over cuttings is that greater genetic diversity will be built into the restoration project. Collecting local seed stocks should also result in fewer negative impacts to fragile wetland ecosystems than the collection of whole plants, plugs, or cuttings.

4.2.3.2 Use of seedlings

We have found the use of seedlings to be an effective method of establishing diverse salt marsh vegetation in smaller restoration projects, and we are currently experimenting with their use on much larger sites. In previous experimental efforts, we have introduced small seedlings with remarkable success. We germinated eight marsh plain species in flats (Figure 4.2; see Section 4.3.2.1) and transplanted 2- to 4-month-old seedlings utilizing three different techniques: (1) 40 × 42 × 4-cm sod blocks (Crown Point project, Box 1.5); (2) 3 × 3-cm cubes cut from flats (Tidal Linkage, Box 1.10); and (3) 5 × 5-cm peat pots (Tidal

Figure 4.2 Six-week old seedlings grown in plastic-lined 40 × 42 × 4-cm flats in the greenhouse at PERL.

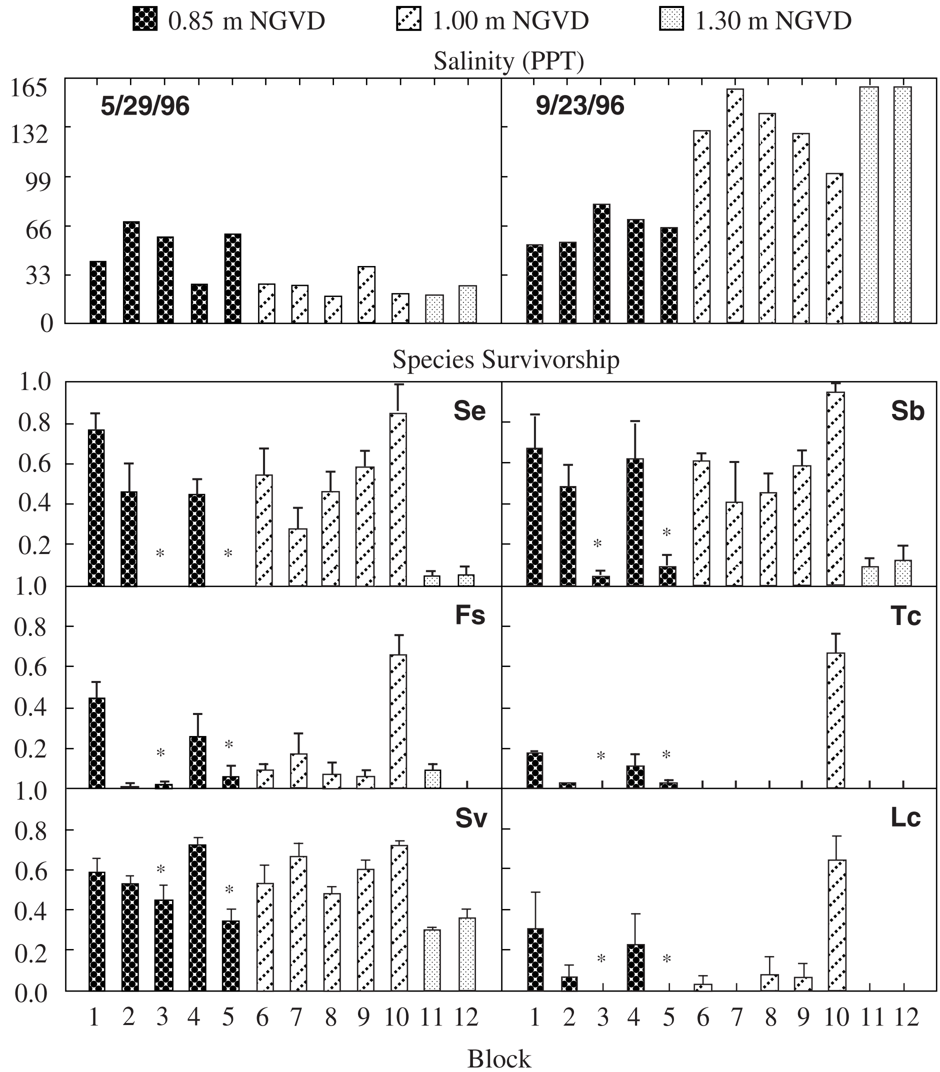

Figure 4.3 Salinity (top) and species survivorship (bottom) within blocks at three elevations at the Crown Point restoration site. Survivorship was measured on November 8, 1996, except for the annual Sb, measured on September 23, 1996. Survivorship was strongly influenced by microsite conditions. Blocks 3 and 5 (*) were poorly drained. Hypersaline conditions developed in blocks 6, 7, 8, 9, 11, and 12 (mean = 146 ppt). Species abbreviations are given in Table 4.4. Means ± SE.

Linkage). In each case, mature plants established within one growing season, with five of the eight species flowering in the first year and the remaining three species flowering in year 2. Seedling survivorship was high — greater than 95% in all but the high density Crown Point sods, where mortality was strongly influenced by the interaction between density and severe microsite conditions (Figure 4.3).

We also tried planting seeds directly onto the marsh surface at Tijuana Estuary, but with mixed success. Broadcasting dry seed worked poorly, as most of the seed floated away on the rising tide. More seed remained on site if seeds were first soaked in freshwater for 24 hours prior to dispersal, but most of these seeds either failed to germinate or also washed away. Establishing seedlings was significantly enhanced by mixing seeds in a soil/organic matter slurry (similar to hydro seeding) that was dropped in small volumes

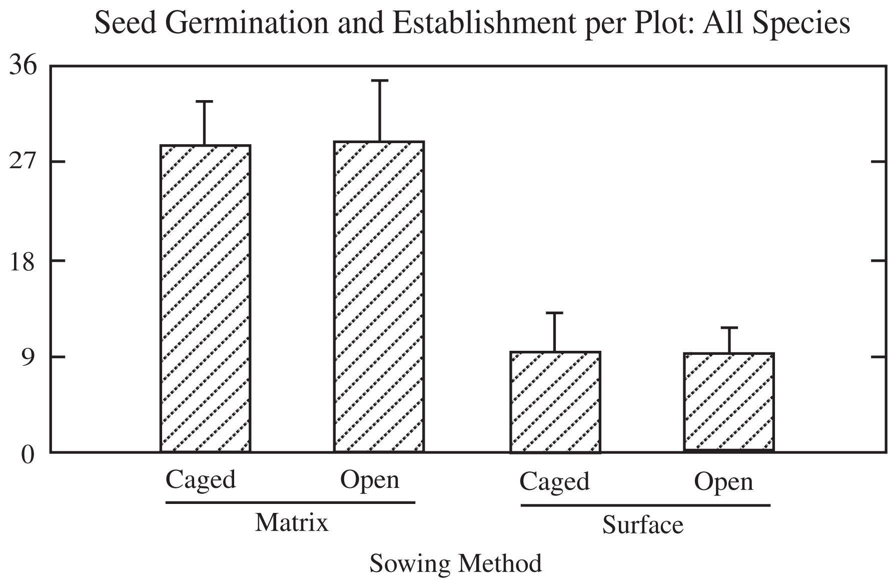

Figure 4.4 Seed germination and establishment of six species at the Tidal Linkage, contrasting sowing methods inside and outside enclosures. Enclosures had no effect. Significantly more seeds established from the slurry matrix than the surface sown presoaked seeds ($P<0.001$). Seeds sown without presoaking floated away.

onto the marsh surface (Figure 4.4). Much of this seed remained within the nutrient-rich material. However, germination of all but the annual species was relatively low, perhaps due to the hypersaline conditions (mean = 68 ppt) found on the marsh surface in the irregularly flooded plots. Irrigation should increase germination and survivorship during the initial establishment phase, although this has yet to be tested (Section 4.4.3.5). For a discussion of other methods, see Burrell (1999).

4.2.3.3 Use of cuttings

Cuttings of several species root and establish readily (PERL, unpublished data). The developing shoots or rhizomes of several perennials will produce adventitious roots when placed in an appropriate rooting medium, after which they may be successfully transplanted (Section 4.3.2.2). Although the relative success of different rooting methods differed among species tested (Figure 4.5), survivorship of all rooted individuals was *100%* six months after transplanting to the Tidal Linkage. This suggests that cuttings will be useful for establishing vigorous plants at low cost, especially for those species not readily available or grown from seed (e.g., *Batis maritima*).

4.2.3.4 Use of sod

Small blocks, cores, or plugs of sod establish very well when introduced to appropriate conditions with a partially intact and functional root system (Gilbert and Anderson 1998; Section 4.3.3). For example, Trnka (1998) introduced *Spartina foliosa* to Crown Point (Box 1.5) using plugs that were 10 to 15 cm wide × 25 cm deep. Transplants of both tall (≥90 cm) and short (≤70 cm) growth forms were collected and transplanted to four plots within the excavated marsh plain in February 1996 and monitored for two growing seasons. The four planting plots differed in sediment quality; each plot received 16 plugs of *S. foliosa*, i.e., 8 tall and 8 short. A 1-m^2 area surrounded by a 0.5-m buffer formed the experimental unit. Year-1 survivorship was uniformly high with little effect of initial height

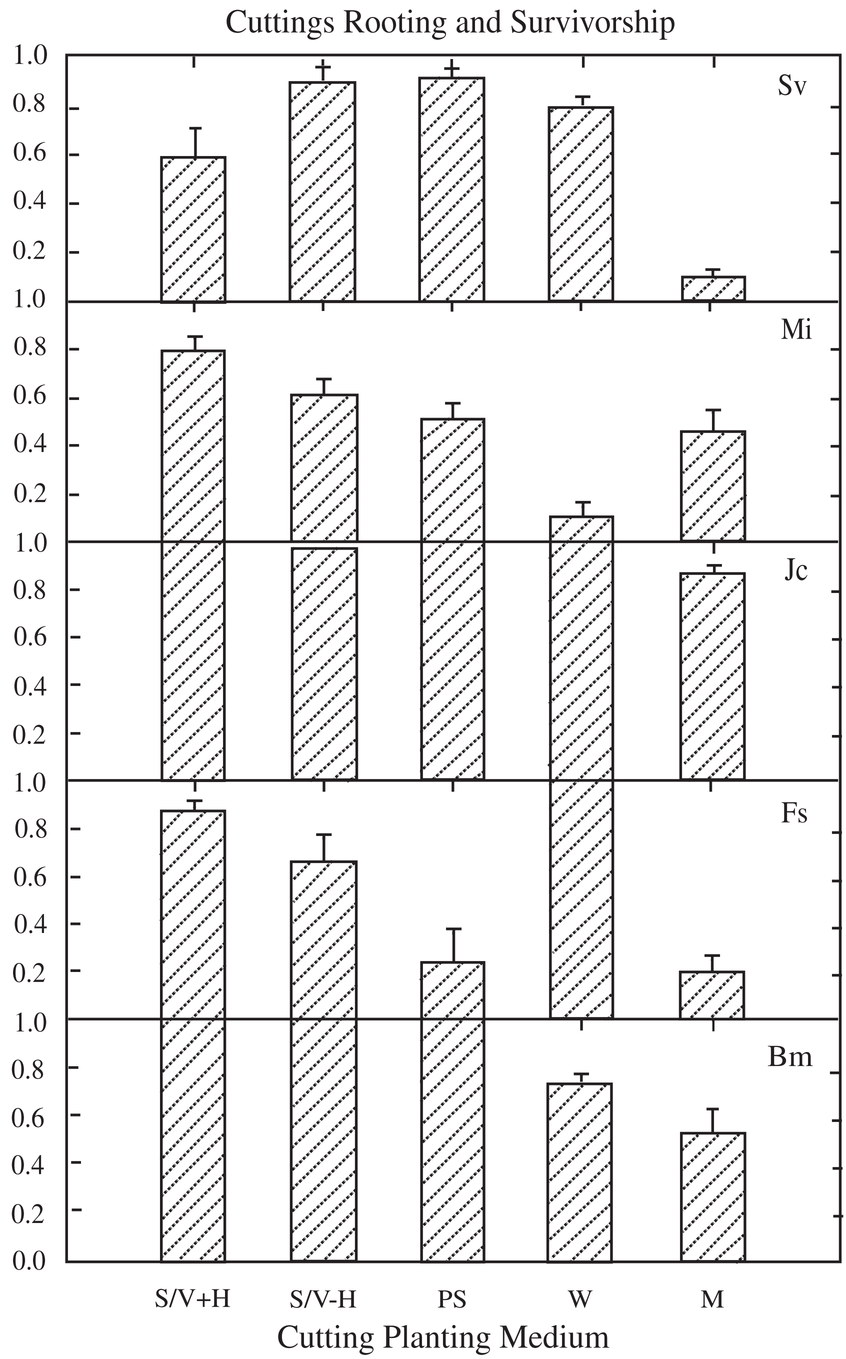

Figure 4.5 Rooting and survivorship of cuttings from five species in four rooting media, plus cuttings planted directly into a salt marsh. All cuttings developing roots survived. Prerooting significantly increased survivorship over cuttings planted in the field. S/V = sand and vermiculite; H = rooting hormone; PS = potting soil; W = water; M = in the marsh.

form on subsequent growth. However, there were significant environmental effects on growth. Clonal expansion rate and the number of stems per clone were lower on sandier sediments, while stems were up to five times more dense and clones up to three times

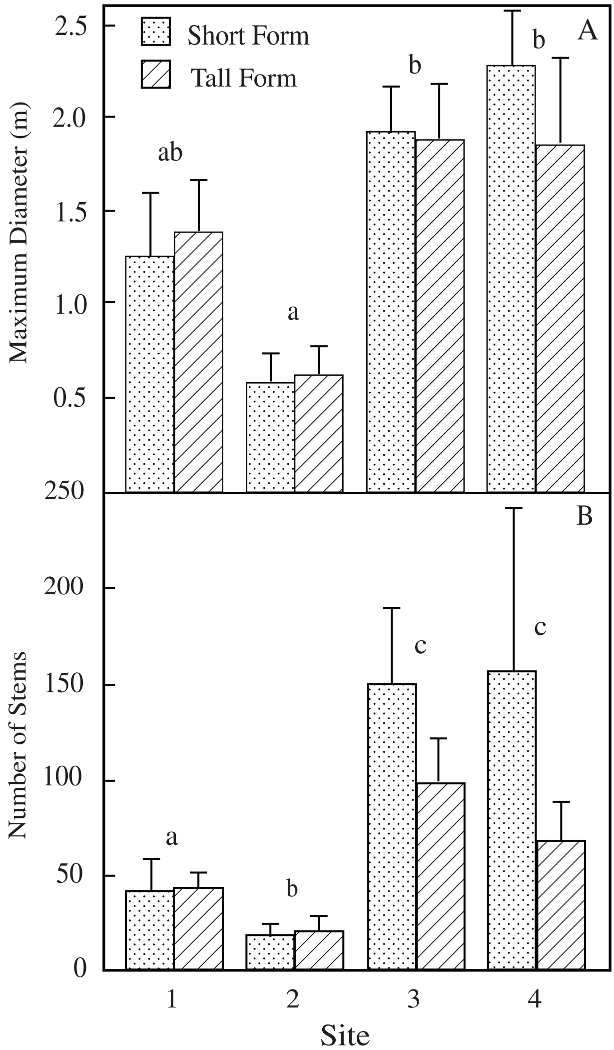

Figure 4.6 Mean diameter of *Spartina foliosa* clones (A) and number of stems per clone (B) grown from 15 cm diameter × 25 cm deep plugs taken from 2 growth forms. Sites 1 and 2 had high sand and low clay content; sites 3 and 4 had higher clay and nutrient contents. Measurements were taken after one year in March 1997. Significant differences among sites are indicated by lower case letters (LSD). Differences between parental growth forms were not significant. Means ± SE. Modified from Trnka 1998, Environmental and parental height form effects on the growth of *Spartina foliosa* in southern California. Master's thesis. San Diego State University, San Diego, CA.

larger on plots with higher clay and nutrient content (Figure 4.6 A, B). There were also more tall stems in the high-clay plots (Figure 4.7). Thus, if vegetative material is available for salvage, the plants can and should be used for restoration. Also, sediment quality at a restoration site can determine how well and how rapidly vegetation will develop — an important consideration for project goals and monitoring strategies.

4.2.4 Summary

Ultimately, the goal of salt marsh restoration is to create a wetland that performs the functions of a natural ecosystem. The planting strategy should be developed to

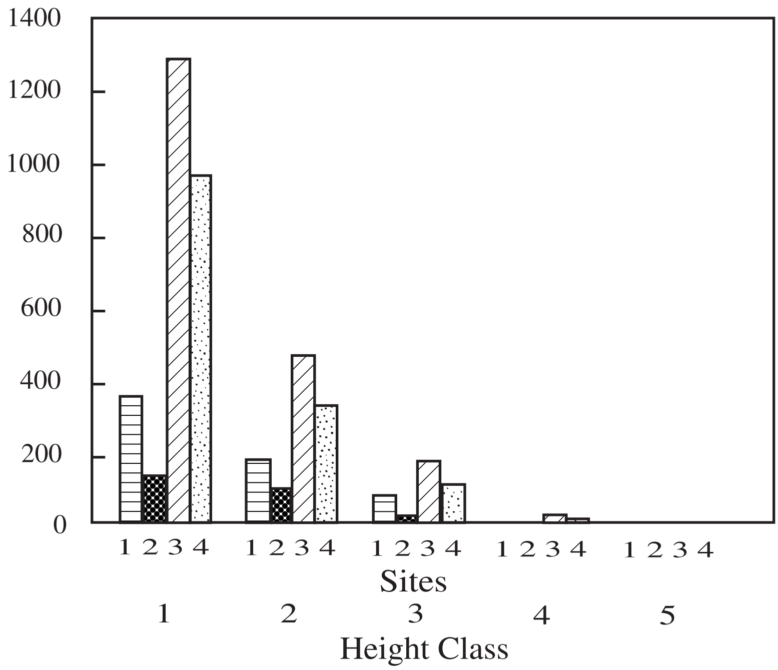

Figure 4.7 Number and distribution of *Spartina foliosa* stems by height classes, after one year at each site. Stem height classes: 1 = 0 to 30 cm, 2 = 31 to 60 cm, 3 = 61 to 90 cm, 4 = 91 to 120 cm, 5 = 121 to 150 cm. Total stem number and height distributions were strongly affected by site conditions (see Figure 4.6). Modified from Trnka 1998, Environmental and parental height form effects on the growth of *Spartina foliosa* in southern California. Master's thesis. San Diego State University, San Diego, CA.

- maximize diversity across the landscape;
- match species with microhabitats, so that developing assemblages are composed of species that naturally interact together on a local scale;
- use the species range and modal elevation as a guide for planting location;
- monitor how the site develops; and
- prepare an adaptive management plan to identify and solve problems.

Local reference systems should be used to:

- build a list of candidate species;
- determine the range and modal elevation of species and their distributions relative to each other; and
- identify species assemblages and their microhabitats.

The source of plant material should be determined for each species well in advance of construction. Planners should:

- contract with a reputable grower to provide the desired plant and genetic stock;
- make arrangements to collect seeds or cuttings with sufficient time for germination and/or rooting prior to outplanting; and
- where applicable, salvage wetland plant (and soil) material from the site to be reused in the construction process.

Finally, the genetic material of plants that will be destroyed in the construction process should not be lost but captured for reuse (Section 4.5).

4.3 Plant acquisition, propagation, and maintenance

4.3.1 Seed collection and storage

Seeds can be collected in substantial numbers as they mature; if done with care, there will be little damage to existing vegetation. Since reproductive phenology differs among species, collections must be made throughout the year. Although the seeds of any one species are available for several weeks, the window of opportunity varies from year to year. The more ephemeral and/or seasonally variable species need to be monitored to determine when seeds are available. This is especially true for species with reproductive cycles timed to seasonal rainfall patterns (e.g., *Amblyopappus pusillus*).

Seed should be collected widely and no more than 5% should be harvested from any one area. This is critical for rare and threatened species, which require seed production and dispersal to ensure population survival. Collecting the seed of federally or state-listed species (e.g., *Cordylanthus maritimus* ssp. *maritimus*) is prohibited without appropriate permits, and collection should not be attempted without carefully considering why it is required and how the seed will be used (Box 2.6). Collecting the seed of any species in state or national parks, wildlife preserves or refuges will generally also require permits, which should be requested well in advance, i.e., during the planning stage.

Once seeds have been collected, they should be air-dried and prepared for storage. Some seeds will need to be separated from maternal structures, such as the inflorescences of *Salicornia bigelovii* (although storing inflorescences intact will not affect the seed). Some seed collections will also include a number of insect herbivores, so precautions should be taken to eliminate them. Drying followed by refrigeration will kill most pests. Other strategies include treating them with insecticide (Young and Young 1986). A refrigeration period also doubles as a cold storage treatment, which may facilitate the germination of many species (e.g., *Spartina foliosa* germinates best after cold storage in freshwater). Once prepared, seeds may be stored for longer periods under refrigeration, although storage in a cool dry place without granivores is generally sufficient. Seed viability over time varies among species and will be promoted with careful preparation and storage practices (see Appendix 2 for notes on seed storage).

4.3.2 Propagation methods

4.3.2.1 Seed

The seed of most species will germinate when given the proper salinity, light quality, photoperiod, temperature, and/or moisture regimes. In southern California salt marshes, most species germinate well with exposure to moist soil at low salinity (Table 4.2), similar to the conditions naturally occurring with seasonal winter rains. This is especially true among the high-marsh species, where germination may occur as early as November or as late as April, depending upon annual rainfall patterns. Temperature and photoperiod also affect the germination of some species, with emergence delayed by shorter days and colder temperatures (Table 4.3). The percentage of seeds germinating for some species may also be improved by an extended period of cold dormancy prior to germination (Young and Young 1986, Emery 1988, Abraham 1991, Deno 1991).

In our experience, the best germination technique for the marsh-plain and high-marsh species is to mix a quantity of seed evenly in loose organic potting soil and then spread the soil/seed mixture in a 0.25 inch (5 to 6 mm) layer over moist topsoil in a flat (Table 4.5). The quantity of seed will be dependent upon the desired seedling density. The organic mixture provides a moist environment and inhibits drying at the soil surface. Germination of the "buried" seed is promoted if the mulch layer is loose, as seeds still experience light

Table 4.2 Proportionate germination of annual species found in high marsh habitats as a function of salinity and soil moisture. Germination of 25 seeds was assessed after 30 days in pots in the greenhouse and growth chamber. Salinity and moisture levels are for the soil surface (top 1 cm). Data from Noe 1999, Abiotic effects on the annual plant assemblage of southern California upper intertidal marsh: does complexity matter? Doctoral thesis. San Diego State University, San Diego, CA.

	Salinity (ppt)				Percent moisture		
Greenhouse Experiment	**2**	**7**	**15**	**31**	**35%**	**41%**	**45%**
Amblyopappus pusillus	0.53	0.44	0.23	0.01	0.22	0.28	0.40
Cotula coronopifolia	0.70	0.61	0.60	0.35	0.53	0.53	0.64
Hutchinsia procumbens	0.18	0.09	0.04	0.00	0.03	0.04	0.16
Juncus bufonious	0.29	0.20	0.14	0.00	0.09	0.16	0.23
Lasthenia glabrata ssp. coulteri	0.27	0.19	0.04	0.00	0.07	0.13	0.18
Lolium multiflorum	0.89	0.94	0.93	0.57	0.76	0.83	0.91
Lythrum hyssopifolium	0.63	0.63	0.54	0.08	0.41	0.47	0.53
Mesembryanthemum nodiflorum	0.33	0.27	0.27	0.00	0.20	0.24	0.22
Parapholis incurva	0.92	0.88	0.88	0.27	0.68	0.72	0.81
Polypogon monspeliensis	0.96	0.97	0.93	0.76	0.87	0.90	0.94
Spergularia marina	0.89	0.88	0.81	0.24	0.65	0.72	0.75
Growth Chamber Experiment	**2**	**7**	**13**	**23**	**37%**	**46%**	**51%**
Amblyopappus pusillus	0.61	0.59	0.48	0.31	0.44	0.54	0.53
Cordlyanthus maritimus	0.70	0.64	0.58	0.46	0.38	0.64	0.78
Hutchinsia procumbens	0.73	0.48	0.24	0.06	0.31	0.39	0.43
Lasthenia glabrata ssp. coulteri	0.84	0.73	0.46	0.12	0.44	0.52	0.66
Mesembryanthemum nodiflorum	0.35	0.31	0.36	0.24	0.29	0.34	0.32
Parapholis incurva	0.99	0.98	0.98	0.91	0.94	0.98	0.97
Spergularia marina	0.71	0.75	0.75	0.47	0.54	0.75	0.73

Table 4.3 Proportionate germination of 8 annual species found in high marsh habitats as a function of temperature and photoperiod. Germination of 25 seeds was assessed after 30 days in pots in a growth chamber. For temperature, 16.7 = Nov. mean; 15.3 = Mar. mean; 21.1/12.2 = Nov. 15 diurnal; 19.1/11.6 = Mar. 15 diurnal temperatures. For photoperiod, 10.5 hours light = Feb. 2 and Nov. 15; while 12 hours light = Mar 15 and Sep. 30. Data from Noe 1999, Abiotic effects on the annual plant assemblage of southern California upper intertidal marsh: does complexity matter? Doctoral thesis. San Diego State University, San Diego, CA.

Temperature °C	16.7	16.7	15.3	21.1/12.2	19.1/11.6
Photoperiod Hours Light	10.5	12	12	12	12
Cotula coronopifolia	0.64	0.76	0.48	0.59	0.63
Hutchinsia procumbens	0.57	0.58	0.54	0.73	0.72
Lasthenia glabrata ssp. coulteri	0.81	0.87	0.85	0.62	0.86
Lythrum hyssopifolium	0.41	0.72	0.31	0.59	0.46
Mesembryanthemum nodiflorum	0.32	0.27	0.30	0.30	0.26
Parapholis incurva	0.82	0.95	0.94	0.63	0.86
Polypogon monspeliensis	0.93	0.96	0.99	1.00	0.98
Spergularia marina	0.66	0.81	0.62	0.58	0.65

cues. Once seeds germinate, they root easily in the organic layer and respond to low concentrations of fertilizer, which may be added if not already present in the potting soil. In greenhouse tests of 8 marsh-plain species, germination was greater using this technique

Table 4.4 Halophyte species of the southern California salt marsh and their abbreviation codes.

	Cordgrass Marsh		
Spartina foliosa	Sf		
	Marsh Plain Species		
Batis maritima	Bm	*Monanthochloe littoralis*	Ml
Cuscuta salina	Cs	*Salicornia bigelovii*	Sb
Distichlis spicata	Ds	*Salicornia virginica*	Sv
Frankenia salina	Fs	*Suaeda esteroa*	Se
Jaumea carnosa	Jc	*Triglochin concinna*	Tc
Limonium californicum	Lc		
	High Marsh Species		
Atriplex watsonii	Aw	*Salicornia europaea*	Seu
Cordylanthus maritimus	Cm	*Salicornia subterminalis*	Ss
Cressa truxillensis	Ct	*Suaeda calceoliformis*[1]	Sc
Lasthenia glabrata ssp. *coulteri*	Lg		
	Freshwater Transitional Species		
*Atriplex triangularis**	At	*Lythrum hyssopifolium*	Lh
*Cotula coronopifolia**	Cc	*Polypogon monspeliensis**	Pm
Juncus acutus	Ja	*Rumex crispus*	Rc
Juncus bufonius	Jb		
	Upland Transitional Species		
Amblyopappus pusillus	Ap	*Limonium ramosissimum**	Lr
Atriplex californica	Ac	*Mesembryanthemum crystallinum**	Mc
*Atriplex semibaccata**	As	*Mesembryanthemum nodiflorum**	Mn
*Bassia hyssopifolia**	Bh	*Parapholis incurva**	Pi
Eriogonum fasciculatum	Ef	*Sonchus asper**	Sa
Frankenia palmeri	Fp	*Sonchus oleraceus**	So
Heliotropium curassavicum	Hc	*Spergularia macrotheca*	Smc
Hutchinsia procumbens	Hp	*Spergularia marina*	Smr
Isocoma venata	Iv	*Suaeda moquinii*	Sm
*Lolium multiflorum**	Lm	*Suaeda taxifolia*	St
Lycium californicum	Lyc		

* = exotic.

1. See Appendix 2.

than for seed sprinkled on the soil surface or buried shallowly in soil or a layer of mulch (Table 4.5). Seedlings established this way also grew quickly with low mortality.

Once established, seedlings can be easily transplanted into individual pots or matured in the flats until ready to plant (Figure 4.2). Seedlings in flats can be cut into cubes for transplanting into pots or directly into a restoration site. This is an economical method of producing marsh-ready plants, with a standard 16 × 17-inch (40 × 42-cm) flat producing over 180 3 × 3-cm cubes. If larger plants are desired for transplanting, they should be planted at lower density. The growth response at various planting densities differs among species, but in general, the growth of seedlings on 3 to 5-cm centers (on average) will be stunted little over 2 to 3 months of growth prior to outplanting. The growth of plants on centers <3 cm will eventually be restricted by intraspecific competition, although these plants may still establish well once transplanted into the field at low densities. Dense plantings (<2-cm centers) work well for germinating large numbers of plants, but they will need to be transplanted or thinned after 5 to 6 weeks.

Table 4.5 Seed germinations per 0.25 m^2, contrasting four different sowing methods over sandy topsoil (sand:silt:clay = 0.73:0.17:0.13) in two greenhouse experiments. Seeds were sown across the surface, raked in 5 mm, mixed in mulch, and spread over the surface 5 mm thick, or sown on the surface and covered with 5 mm mulch. Mulch was a commercial organic potting soil (A1 Potting Soil). In both experiments, germination of all species was significantly greater when seeds were mixed with mulch. There was no difference between seeds raked into the soil and those sown on the surface in Experiment 1. Sq (*Salicornia quintin*), is what we are calling a previously undescribed species found on San Quintin Bay, Baja California Norte, Mexico. See Table 4.4 for other species abbreviation codes.

Experiment 1	All	Bm	Fs	Jc	Lc	Sb	Sq	Se	Sv	Tc
on surface	380.4	0.0	4.0	1.3	12.1	44.4	215.1	—	49.7	53.8
raked in	275.5	1.3	8.1	1.3	9.4	17.5	204.3	—	16.1	17.5
mixed in mulch	884.4	18.8	119.6	30.9	90.1	84.7	392.5	—	9.4	138.4
Experiment 2										
mixed in mulch	451.6	—	85.2	—	5.8	70.8	—	23.3	179.5	87.1
under mulch	227.6	—	42.6	—	3.2	32.1	—	18.5	81.8	49.5

4.3.2.2 Cuttings

Many species can be propagated from cuttings, that is, from cut stems developing adventitious roots from meristematic tissue, usually at nodes along the basal end of the cut shoot (Hartman and Kester 1983). Different techniques may be used, depending upon the species characteristics. From our experience with halophytes, species that root best from cuttings are generally those with aboveground aerial stems or with shoots developing from stolons or rhizomes. Rosette species and bunch graminoids are poor candidates, although some species may develop new shoots from leaf cuttings (Browse 1988). Some species (e.g., *Monanthochloe littoralis*) can develop new shoots from any node, so that a single branch or runner may give rise to many individual plants.

The time of year that cuttings are collected may be critical to rooting. Cuttings should be taken from actively growing herbaceous stems, not woody or dormant plants. Plants with reproductive shoots should be avoided when possible, because the plant is not in vegetative growth mode, although such shoots may still produce roots. Stems approximately 10 to 12 cm in length should be collected and kept in a cool damp place until ready for planting, which should take place as soon as possible after collection. If shoots must be stored for short periods of time, cut ends should be placed in water or the entire shoot kept in a sealed plastic bag with enough water to maintain 100% relative humidity. Shoots should be planted in a rooting medium with at least one-third the length of the stem and at least one node below ground. Excess shoot material will increase transpirational water loss and decrease survivorship. Cuttings of stolons or rhizomes should be planted with the runner below ground and any shoots developing from nodes extending above the soil surface.

Because cuttings can be rooted in a number of moist environments, including water alone, we tested the rooting potential plus subsequent growth and establishment of several marsh-plain to high-marsh species planted in the greenhouse and in the field (Figure 4.5). *Salicornia virginica*, *Frankenia salina*, *Jaumea carnosa*, *Monanthochloe littoralis*, and *Batis maritima* cuttings were tested in five rooting media:

1. 1:1 mixture (volume:volume) of sand/vermiculite with rooting hormone applied to the cutting (basal tip dipped in Hormex™ standard strength rooting powder-IAA),
2. 1:1 mixture (volume:volume) of sand/vermiculite without rooting hormone applied,
3. commercial organic potting soil (A1 Potting Soil),
4. tapwater (with no nutrients added), and
5. planting directly into plots at the Tidal Linkage.

Cuttings at the restoration site were planted with at least one node and approximately 50% of the stem below ground. Elevation of the restoration site plots was 0.9 m NGVD (= 5.4 feet local MLLW; see Section 6.2.6.1), where they received tidal flushing at least once a day for 31 days of the 60-day experiment without irrigation.

Some cuttings rooted in each treatment, but there were significant differences among rooting media and species (Figure 4.5). Overall, survivorship after 60 days in potting soil and both sand mix treatments was identical (approximately 84%) and significantly greater than in water (73%) or marsh plantings (43%). The use of rooting hormone had little beneficial effect among these species, and significantly inhibited root development in *Salicornia virginica*. The data suggest that cuttings of all five species root well in a soil treatment, but neither the use of rooting hormone nor the more expensive potting soil significantly improved survivorship. *Jaumea carnosa* rooted exceptionally well in all treatments (survivorship >95%), exhibiting an average growth rate (percentage weight gain) 3× that of the next best species. All species rooted fairly well in water alone except *Monanthochloe littoralis*, which had extensive mortality. *Suaeda esteroa* and *Salicornia subterminalis* were also tested in the sand mix without hormone. All of the *S. esteroa* developed roots and established well, but only about 20% of the *S. subterminalis* rooted and survived until planted in the field.

Three of the species rooted well when simply planted on the marsh plain in an area receiving irregular tidal inundation (no flooding for 31 of 60 days). *Jaumea carnosa*, *B. maritima*, and *M. littoralis* should each establish well under similar or better conditions. Although *S. virginica* and *F. salina* did not do well when planted directly into the marsh, we believe that their survivorship and that of most species could be substantially improved by planting cuttings with up to two-thirds of the stem and all of the rhizome below ground. Survivorship will be substantially enhanced with irrigation, especially where cuttings receive little or intermittent inundation.

Survivorship was *100%* for all rooted cuttings transplanted into the Tidal Linkage. However, those cuttings rooted in a soil treatment were generally more robust and flowered earlier than those rooted in water. These data suggest that the use of rooted cuttings is a viable method for establishing vegetation on newly created or restored sites, especially cuttings of species that establish poorly from seed such as *B. maritima*. Using cuttings of species that readily establish when planted directly into the site, such as *J. carnosa*, may also prove to be far more economical than purchasing potted plants. Used in conjunction with other techniques, vegetating with cuttings should be a valuable tool to marsh restorationists.

4.3.3 *Salvaging plants with sod*

Salt marsh vegetation is a valuable resource of those coastal wetlands still subject to habitat loss or degradation in southern California and elsewhere. Consequently, every opportunity should be taken whenever practical and economically feasible to salvage plant material that will otherwise be lost or destroyed in this process (Zedler 1996). Salvaged material requires immediate processing or use (e.g., seeds should be prepared for storage and cuttings should be rooted or planted quickly; see above), although cores, plugs or sods can be stored and maintained for months or years with appropriate care.

4.3.3.1 *Salvage methods*

Perhaps the best method of salvaging plants is the collection of whole plants and the soil in which they are rooted. Plants plus a portion of the marsh soil and its biota can be salvaged by taking plugs or cores of sod, while larger expanses of the marsh community may be collected relatively intact as sod blocks (Pywell et al. 1995). This offers the significant

advantage of small or negligible transplant shock for the plants — because the roots, and especially the fine root hairs, remain relatively undisturbed in their local soil environment. Such plants recover more quickly and begin growing more rapidly in an intact and functional root environment. An important benefit of transplanting the local root environment is that plants are introduced in a "cuvette" of natural marsh sediment that is usually of higher quality than soils found in the created site (Chapter 3). Establishing plants from plugs or sods often means introducing a more diverse interactive community — plugs rarely contain a single plant or species, but include a multi-species turf of shoots, roots, and rhizomes, with associated microbial and invertebrate components (Shisler 1990). Establishing diverse assemblages should improve chances of obtaining self-sustaining ecosystems, and naturally functioning assemblages.

Extensive areas of high quality wetland may be salvaged by excising plugs, cores, or blocks of sod for temporary or long-term storage prior to their reintroduction to a created marsh. Such sods can be transplanted entirely into a created wetland or subdivided into many smaller blocks at planting time. Although the vascular plant community can be maintained as a block of sod with irrigation and care, other elements of the marsh community may well be lost in this process, especially when the sod is stored for longer periods. Bacteria, fungi, protists, and invertebrates rely on the tides for a variety of functions including transport, detrital and nutrient inputs, oxygen exchange, and general environmental regulation. Freshwater irrigation can leach out salts and allow weedy exotics to colonize the stored sod blocks. If stored too long, the native vegetation may be lost under such conditions. The storage conditions will differ radically from the dynamic marsh environment, especially for organisms found at lower elevations. More research will be required before we can predict the effects of storage on the entire community, or what might be done to promote overall survival. In the meantime, we advise that salvage efforts be carefully planned, storage periods minimized, and actions taken to reduce temperature, salinity, and soil moisture fluctuations.

Heavy equipment (e.g., Bobcat or Payloader) can cut relatively large 20 to 30-cm deep chunks of sod, while smaller blocks can be cut by hand (Figure 4.8A). Although deeper blocks are heavier and more bulky, the additional roots and high quality soil will strongly benefit the plants after replanting. Salvaging whole sod blocks works best with soils that maintain their structural integrity after being 'cut' from the ground, typical of the high clay content soils found in natural marshes (Figure 4.9). Timing sod harvest with the tide cycle will be critical to this process, so that soil moisture levels are appropriate for cutting sods and holding them together, i.e., overly wet sod may be too loose or runny, while dry sod may be too hard or granular. A dense root zone and highly developed turf help blocks retain their integrity. However, sods cut from sandier soils easily fall apart under manipulation and/or subsequent irrigation. If the sod blocks break up, root connections and fine root hairs will be damaged, microbes exposed to the air, and mycorrhizae lost. At the other end of the spectrum, extremely wet and/or loose sediments may also break up. In both of these circumstances, the soils may still be salvaged by carefully cutting sections, cores, or plugs and keeping them in 1- to 5-gallon containers.

Before salvage efforts begin, the ground should be opened at the edge of the area to be salvaged; otherwise, equipment and foot trampling will severely damage the marsh vegetation. Whatever the block size, the sides and back face should first be cut to the desired depth, or the roots will tear and the soil structure will be damaged as the block is being removed. Once the vertical faces have been cut, the bottom should be cut with the blade used to lift and remove the block (e.g., a hand shovel, front-end loader bucket, or customized sod cutting tool). This way, blocks can be systematically removed, like pieces of cake, and transported to a holding area (Figure 4.7). Smaller blocks can be condensed and moved on pallets for short holding periods. These salvage techniques were

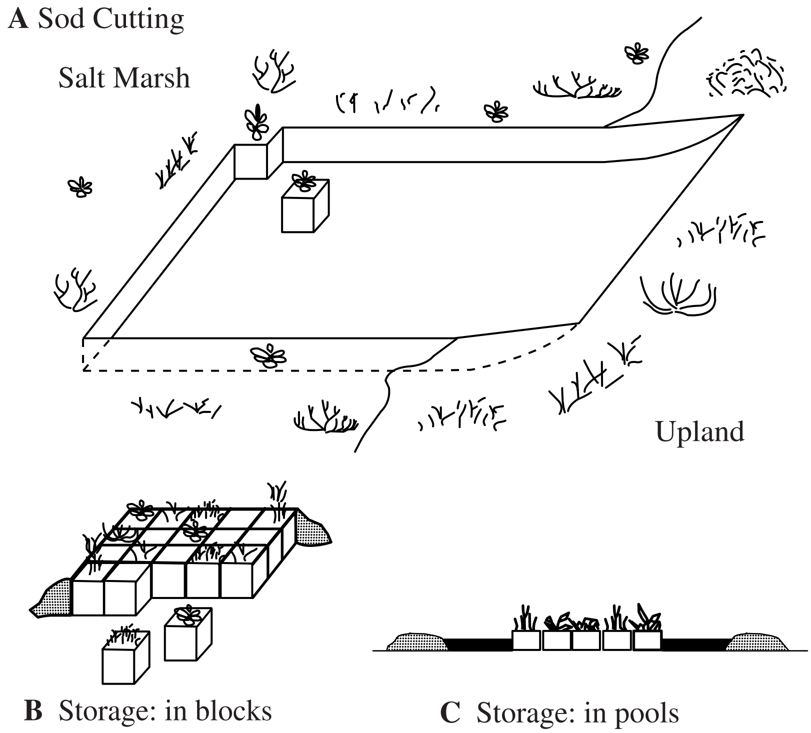

Figure 4.8 Salvage of sod for long- or short-term storage prior to reuse. Smaller blocks are cut by hand, while larger blocks are cut with heavy equipment. Salvaging begins at the marsh edge (A). Blocks should be stored side by side, with edges and joints sealed with loose soil (B). Blocks may also be stored in pools, in flats, or pots (C).

used effectively during construction of the Tidal Linkage (Box 1.10) in 1997. Moreover, they can be employed wherever remnant marsh vegetation can be found on a restoration site.

4.3.3.2 Storage and maintenance during construction

To store sods for more than a day or two, blocks should be placed tightly side by side, with additional soil or sand mounded around the edges of the array to reduce evaporative loss from the exposed root zone (Figure 4.8B). Salt marsh sods collected from higher elevations can be kept on bare ground for short terms so that irrigation water does not impound and stress the plants. For longer storage, they should be stored on a polyethylene sheet to prevent weedy invasions from adjacent or underlying vegetation. Sods collected from areas frequently inundated or saturated should be kept in shallow depressions with a polyethylene liner to retain water and prevent excessive desiccation (Figure 4.8C). Five to 10-cm deep water should be sufficient for sods 20 to 30-cm deep. A layer of sand over and under the liner may be needed to prevent puncturing by roots or sharp objects in the sod or on the ground.

Long-term storage of sods requires additional precautions. Because sediment salts are readily leached by freshwater irrigation, especially with sandier soils, occasional additions of salt water will be necessary to maintain higher salinities. This will require periodic monitoring. Plants held in nonleaking structures should only need salt watering once, with freshwater additions to replace evaporative loss. Otherwise, low salinity may encourage the growth of freshwater species, exotics, or invasive upland weeds in the seed bank,

Figure 4.9 Profile of the rooting zone at Tijuana Estuary after 40 cm of marsh plain sod was removed with a Bobcat front-end loader. Blocks of sod (approximately 40 × 40-cm × 40-cm deep) were salvaged from the new channel basin, stored on site, then cut into smaller blocks (approximately 10 × 10-cm × 30-cm deep) to replant the newly excavated marsh plain. One m^2 of sod planted approximately 50 m^2 of the restoration marsh plain on 1-m centers.

any of which may outcompete native salt marsh species once they become established (Zedler 1991, 1996, Kuhn and Zedler 1997, Callaway and Zedler 1998).

4.3.3.3 Holding ponds

Although salt marsh plants may survive and grow well following storage, the magnitude and direction of interspecific interactions may be altered, so that some species flourish at the expense of others. We examined plant response to long-term storage among cores or sods collected from high marsh, marsh plain, and cordgrass habitat.

Sods were held in polyethylene-lined holding ponds (5-cm water depth) and monitored for 7 months. Seawater was added to the ponds to raise salinity to approximately 17 ppt, with evaporative water losses replaced with freshwater. In the first study, 10-cm-square cores of high marsh sod dominated by either *Salicornia subterminalis*, *Frankenia salina*, *Monanthochloe littoralis*, or *Distichlis spicata* were collected at depths of 10 or 20 cm and kept in plastic pots. The overall survivorship of target species was very high, with core depth only affecting the survivorship of *S. subterminalis* (a deep rooting species) and the growth (in height) of *F. salina* (Figure 4.10). However, all cores suffered a dramatic invasion of exotic annuals in the sandy high marsh soils (chiefly *Parapholis incurva*). Aggressive exotics will slow growth of the desired natives. Although collecting deeper cores will salvage more root and marsh soil, salt marsh plants can be effectively salvaged in shallow sod or cores, especially species lacking deep root systems. Shallower cores may be desirable for salvaging marsh vegetation growing in relatively low quality sediment that provides little benefit to a restoration site. In high marsh areas, irrigation will be needed until deeper roots develop. This study suggests that higher salinities should be

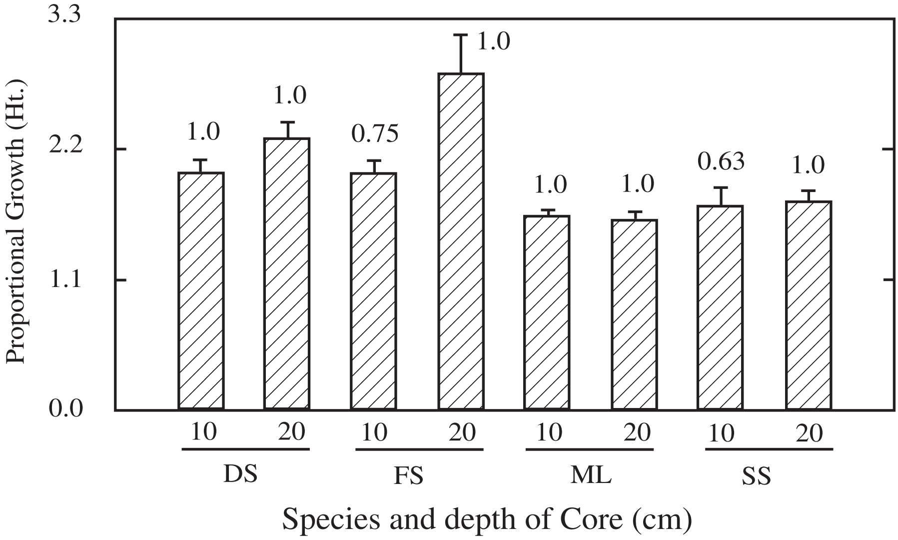

Figure 4.10 Growth and survivorship of four high marsh species after transplanting cores into 1-gallon pots for 7 months. Differences were assessed between 10- and 20-cm deep cores. Growth of survivors was measured as the proportionate increase from initial height (final/initial ht.). The effect of core depth was significant only for *F. salina.* Survivorship is given above each bar.

maintained to prevent the germination and establishment of exotic competitors in the seed bank.

In the second study, sixteen 40 × 40-cm sod blocks cut to 10 and 20 cm depth were maintained in holding ponds on flats. The sod blocks contained varying densities of *Salicornia virginica, Frankenia salina, Suaeda esteroa, Limonium californicum, Triglochin concinna, Jaumea carnosa*, and *Batis maritima*. *Cuscuta salina* also appeared in three of the blocks during the study period. All species except *L. californicum* and *B. maritima* flowered during the study. *S. esteroa* and *L. californicum* did not survive in the shallow blocks over the 7 months, but all other species survived at both depths and exotic invasions were few (chiefly *Atriplex triangularis* and *Polypogon monspeliensis*). Based on visual observations, relative densities changed toward the end of the monitoring period. *S. virginica* and *J. carnosa* began to dominate, while *T. concinna* decreased in overall density.

In the third study, we monitored the survivorship of *Spartina foliosa* in multispecies sods maintained in a holding pond environment. In the area of collection, *S. foliosa* roots and rhizomes extended down 25 cm, with the roots of other species (*B. maritima, J. carnosa*, and *S. virginica*) mainly found in the top 10 cm of sediment. Twenty-two 25-cm dia. × 25-cm deep sods were kept in 5-gallon containers. Mortality was relatively low, with *S. foliosa* surviving and flowering in 17 of the 22 assemblages.

After 14 months in the holding pond, sod from each of these studies was transplanted into the Tidal Linkage. Cores were planted whole into an area just above and below the seasonal high tide elevation, while the rectangular sod blocks were cut into six 13 × 20-cm sections before planting on the marsh plain. The *S. foliosa* sods were planted whole adjacent to the channel banks. Survivorship among all of these plantings appeared to be 100%, although some individuals may have died without detection. The exotic annuals did not reappear in those cores receiving inundation on higher high tides, but they did return in cores planted above the high tide line. One year later, all of these plantings had become integrated into the Tidal Linkage salt marsh.

4.3.4 Salvaging soils

One of the most valuable of wetland resources is the soil (Chapter 3). Created wetlands seldom have soil with texture or organic content similar to that found in a natural marsh, and these characteristics take decades to develop (Craft et al. 1988, Gibson et al. 1994, Zedler and Callaway 1999). Fine, organically rich soils also contain some of the critical biotic components of a wetland that may be required before the system can achieve self-sustainability — the bacteria, detritivores, fungi, and mycorrhizae. Additionally, salvaged wetland soils may contain a seed bank that will establish a diverse flora or augment planting efforts, if given the appropriate hydrology, light, and temperature regime (Shisler 1990). Moreover, seed banks can be a valuable source of wetland plants that are preadapted to local biotic and abiotic conditions (Ter Heerdt et al. 1994, Wilson et al. 1993, Grillas et al. 1993). At every wetland area that is to be altered, the existing or historical wetland soils should be set aside for reuse. Even small soil banks can be spread thinly over a newly contoured area, inoculating the ground and providing a skin with desirable density and texture characteristics.

Stockpiling soil has the added benefit of reducing the cost of trucking it off site. Salvaged soil may be kept on site for weeks or months with care. As soils dry, they lose some of their permeability characteristics, especially soils with the finer clays typical of natural wetlands. Soil moisture can be maintained with periodic watering, utilizing a scheme appropriate to the project's scale. For longer storage periods, a polyethylene or other suitable cover will retard evaporative loss. Such covers must be well secured around the entire base to prevent wind exposure or being blown away. If possible, soils should be stockpiled out of direct sunlight or under shade cloth to reduce excessive and potentially lethal temperatures.

Despite these precautions, biotic components of the system may be unavoidably lost in time. Stockpiling that mixes upper and lower sediments into compact piles will reduce gas exchange, and non-mobile soil organisms and the seed bank may not survive long in an anoxic environment subject to temperature fluctuation and/or desiccation stress. Prolonged exposure to oxygen can also lead to acid sulfate soils (Chapter 3), so pH levels should be checked prior to soil reuse. To maintain biota along with the soil textural characteristics, we recommend that the upper 5 to 10 cm of the topsoil horizon be skimmed off and stockpiled separately, with the storage period minimized. Larger projects may be excavated and graded in sections, so that stockpiled soil can be reintroduced quickly and irrigated as appropriate.

4.4 Planting methods

4.4.1 Timing

Marsh planting should be timed to accommodate biological, practical, and economic circumstances. Plans should include enough lead time to collect and propagate the required plant material prior to vegetating the site (Section 4.4.2). Because species flower at different times of the year, seed will need to be collected periodically (between May and December in San Diego County). Cuttings are best collected while the parent material is actively growing and early in the growing season, so individuals have time to develop after transplantation. Whole plants may be taken at any season, but they establish best when collected in late winter or spring just prior to breaking dormancy. Planting to coincide with or precede seasonal rains or spring tides can ensure establishment. Survivorship may be very poor for plants introduced on a neap tide series without irrigation.

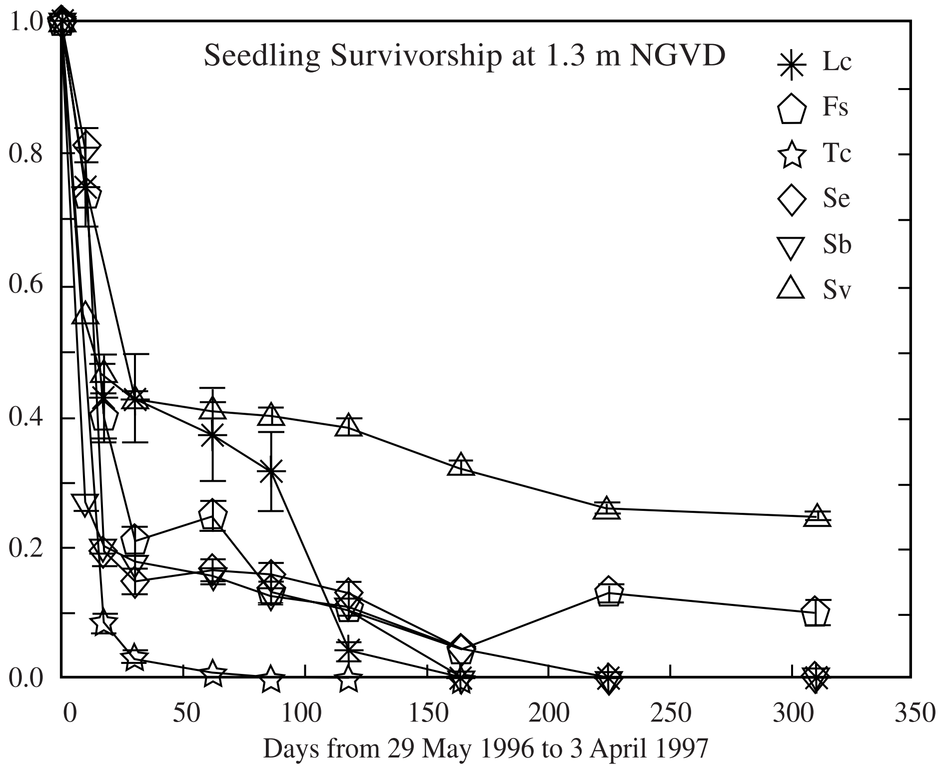

Figure 4.11 Seedling survivorship curves by species at the Crown Point restoration site at 1.3 m NGVD. Species abbreviations are given in Table 4.4. Bars represent ± one standard error.

High mortality (>50%) was recorded at Crown Point (Box 1.5) 10 days after planting at 1.3 m NGVD (Figure 4.11).

Many created wetlands are located adjacent to natural upland or wetland areas. Care must be taken to ensure that the bordering plant and animal community is not harmed by any construction associated with restoration (Josselyn et al. 1990). The use of heavy equipment, plus digging, trucking, and storing materials, may disturb the breeding activities of waterfowl or the nursery use of the marsh surface by juvenile fish. Impacts to rare and endangered species must be avoided. We recommend that careful planning precede the construction process, and that activity be completed prior to the beginning of bird nesting in sensitive areas (as early as Feb. 15th for some California species).

Unfortunately, many circumstances can delay start and completion dates. Contracts may not come through when expected. Money targeted for a project may not be allocated, or it may be withdrawn in whole or part due to political pressures. Contractors may also be unable to proceed on schedule as they fulfill other obligations. Difficult decisions must then be made on when to proceed. Biological expertise is critical here and an adaptive approach is needed (Shisler 1990). All else being equal, ground breaking should be delayed if circumstances dictate a late starting date that might compromise the short- or long-term prospects for a self-sustaining wetland ecosystem.

Once the decision to proceed has been made, the planting effort should take advantage of local weather and tidal patterns. After opening the site to tidal action, spring tides will inundate the area and can leave the soil supersaturated. This not only impairs planting; it can also result in serious disturbance and/or compaction of sediments if workers have to wade through mud. We recommend planting middle and high elevations during neap

tide cycles, to maximize low-water time, and we recommend planting the low elevations during spring tides, when the extreme tides will drain the lower elevations. Seasonal rainfall patterns should also be considered in scheduling. In southern California, planting should be concluded in time for plants to take advantage of winter rainfall and the associated lowered salinities. This is critical for the annuals of the high marsh, which depend on rain. For sites not yet open to tidal flushing, planting should be completed just before the scheduled opening to avoid desiccation stress. Without tidal flushing, a well-designed irrigation scheme is critical. An irrigation system will allow flexibility in planting prior to channel opening (see Section 4.4.3.5).

Failure to consider these factors may be ecologically and economically costly. Potentially irreplaceable plants may be lost, equipment idled, and workers' time wasted as a result of down time. Optimal planting windows may be separated by weeks, and entire projects may be shut down if construction extends into sensitive breeding seasons. Significant delays may necessitate waiting an entire year for the appropriate conditions to become available again. Consequently, scheduling should be dictated by seasonal weather and tidal patterns, the requirements of species to be planted onsite, and avoidance of impacts to rare and endangered species. The need to time contractor activities around seasonal processes cannot be overemphasized.

4.4.2 Preparing plants for the salt marsh environment

The logistics of excavation and grading and the potential addition of soil amendments may dictate that the planting strategy be implemented in stages. This should include the finish grading, contouring of tidal creeks, and the construction of fences to keep out unwanted visitors (Chapter 7). Although the goal is to create a self-sustaining system, created marshes are initially fragile and may be susceptible to minor stress or perturbations, such as desiccation, herbivory, or trampling by dogs or the public. A number of factors should be considered before plants go into the ground. These include species choice and transplant size, preplanting conditioning, microsite targeting, planting densities, and a postplanting irrigation scheme.

4.4.2.1 Plant size

Plants can be transplanted at any stage, from seedling to mature plant. Larger plants have a broader canopy and root system, enhancing their ability to compete during early establishment. A larger root/soil volume will also help buffer the plant from transplant shock and stressful conditions. For example, deeper roots allow plants to avoid the harsh surface environment that can develop without irrigation on newly created sites (Figure 4.3). We found high mortality among shallow-rooted seedlings after only 14 days at Crown Point (Box 1.5, Figure 4.11), due to desiccation stress on the dry hypersaline soils. Although not directly tested, we believe that larger, more deeply rooted plants will have higher survivorship under drier conditions. On the other hand, larger plants take more time to grow, cost more individually, require more handling, and can potentially limit the total number of plantings.

Small plants are ready sooner and require fewer resources (less greenhouse space, smaller pots, less soil, and lower maintenance per plant). Our research suggests that the survival of small plants grown from seed can be very high (Figure 4.1). Seedlings 10 to 14 weeks old transplanted early in the growing season can attain reproductive size in the first year (Table 4.6). Cuttings also root best when initially small, and they may show vigorous growth in a few weeks to months. Survivorship of rooted cuttings transplanted at the Tidal Linkage was 100%, with four of the five species reproductive by the end of the first growing season (Table 4.6). Plants introduced from plugs or sods of any size

Table 4.6 Reproductive status (percent classes) of species transplanted into the Tidal Linkage at Tijuana Estuary after 1 and 2 growing seasons. (A) seedlings without soil amendments, (B) seedlings with organic soil amendments, and (C) cuttings without soil amendments. Species abbreviations are given in Table 4.4.

	Year 1				Year 2			
Species	none	<10	10–90	>90	none	<10	10–90	>90
A								
Bm	X						X	
Fs		X					X	
Jc		X						X
Lc			X					X
Sb				X				X
Se			X					X
Sv				X				X
Tc		X					X	
B								
Bm	X						X	
Fs				X				X
Jc				X				X
Lc				X				X
Sb				X				X
Se				X				X
Sv				X				X
Tc				X				X
C								
Bm		X					X	
Fs			X					X
Jc				X				X
Ml	X					X		
Se	X							X
Sv		X					X	

should do well as long as the root system is intact. Considering their potential for rapid growth, high survivorship, and early reproduction, small plants or seedlings are a good choice for restoration. To establish diverse, regionally adapted vegetation within a 5-year period, we recommend the use of large numbers of smaller plants in conjunction with locally salvaged plants, sods, and/or cuttings whenever they are available.

4.4.2.2 Salt hardening

The salt tolerance of halophytes needs to be induced, so plants must be conditioned to the salinity of the field environment before planting. Physiologically, the plants' internal osmotic pressure must be higher than that of the external environment in order to take up water. The osmotic pressure within plants cultured with freshwater is too low to extract water from a saline environment. Although osmotic strategies differ among species, each requires a gradual increase in external salinity to avoid "osmotic shock." We have found that watering with dilute seawater works best, beginning with a 1 in 8 dilution (approx. 4 ppt), and increasing that by 4 to 6 ppt every 3 to 4 days. Seedlings may acclimate more quickly, and they can be brought up to 35 ppt in about 10 days.

Exposure to direct sunlight increases evapotranspiration and induces salt stress more quickly, requiring a more gradual increase in the watering salinity. If plants are maintained with saltwater for an extended period of time prior to planting, watering with full strength seawater will quickly result in hypersalinity. With construction delays, plants may be held for long periods but must be ready to plant when the opportunity arises. Once they are acclimated, many halophytes can tolerate salinity twice that of seawater, but watering periodically with freshwater ameliorates salt buildup, and plants can be held in good condition indefinitely. Salt crystals (sea or rock salt) can also be sprinkled onto the soil surface and watered with freshwater, but salinity is difficult to control, and this technique can lead to hypersalinity before plants have acclimated.

One final consideration is the soil salinity at the planting site. Soils previously exposed to tidal influence may already be hypersaline (Zedler 1996). Salinity may decline once tidal flushing is restored, but irrigation with freshwater prior to planting may be necessary to prevent severe transplant shock or desiccation stress (Section 4.4.3.5). Survival of plants introduced to hypersaline areas can also be improved by conditioning them for a longer period of time to the salinities found at the transplant site.

4.4.2.3 Sun hardening

Plants raised in greenhouses or under shade cloth must be gradually acclimated to full sunlight or they may suffer tissue damage. The development of accessory pigments (flavonoids), thought to protect cellular organelles and DNA from short-wavelength radiation, needs to be induced (Wong 1976, Salisbury and Ross 1986). Sun hardening can take place over one to a few days by exposing well-watered plants to direct sunlight for one to a few hours, and then covering them with shade cloth or moving them back indoors. Under intense sunlight, plants should be initially exposed for shorter intervals, with exposure spread out over several days. However, under overcast skies, plants can often acclimate without special protection.

4.4.2.4 Fertilization

Fertilizers can stimulate plant growth and root development, significantly shortening the time it takes to grow vigorous and healthy plants large enough for transplanting. However, heavy fertilization can decrease root:shoot ratios through increased canopy development (Fitter 1986, Fitter and Hay 1987). Once plants with a dense canopy are transplanted, transpiration rates might exceed the root system's ability to take up water, especially in dry or hypersaline soil. Also, fertilizer use can increase plant tissue nutrient concentrations, potentially attracting herbivores or facilitating infection by fungi and parasites (Mattson 1980, Boyer and Zedler 1996; see Chapter 7).

We recommend using fertilizers early during propagation to stimulate plant growth. A soluble general-purpose fertilizer or Hoagland's solution works well for seedlings (Broome 1990). We have used soluble urea in nitrogen-poor soils, but a balanced N-P-K fertilizer will work well under most circumstances. Concentrations needed will depend on the method and frequency of application, so follow the recommendations of the manufacturer. Concentrations for cuttings or seedlings should initially be halved, then increased gradually to full strength after the plants begin to grow. Suspending fertilizer use 30 to 45 days before planting may help stimulate root development. For plants that have grown to the planting size and are being maintained outdoors for an extended period, we recommend using fertilizers sparingly; this will slow growth and discourage exotic plant establishment. Fertilizers should be applied when plants exhibit signs of nutrient deficiency, applying just enough to ameliorate symptoms.

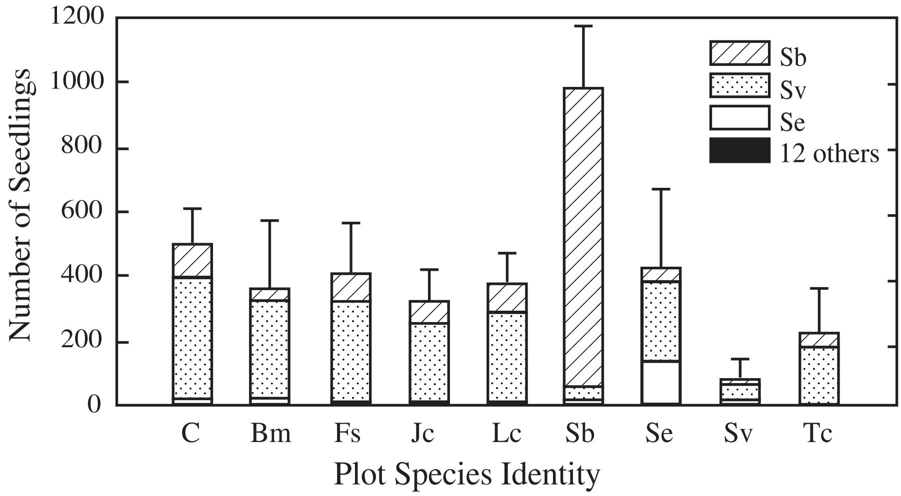

Figure 4.12 The number of seedlings in single-species plots at the Tidal Linkage after 1 year. Fifteen species established seedlings, over 95% of which were Sv, Sb, or Se. The other 12 were Tc, Jc, Lc, Fs, Bm, Sf, Mn, Pi, Ml, Bh, At, and Sm. Species abbreviations are given in Table 4.4. C = control plot.

4.4.3 Outplanting

4.4.3.1 Species relative density

Although we strongly advocate planning for diverse wetlands, it may not be advisable to plant all available species at the same density. Some species are aggressive colonizers that will establish readily on their own. Meeting cover requirements by densely planting aggressive species (e.g., *Salicornia virginica*) is of questionable long-term value because such species can prevent the establishment of slower-growing, less-aggressive species (e.g., *Triglochin concinna*) (Ewel 1990). For example, of the 15 species that recruited from seed at the Tidal Linkage (Box 1.10), three species established more than 95% of all seedlings, with the vast majority either *S. virginica* or *S. bigelovii* (Figure 4.12). *S. virginica* was the highest recruiter in all but the *Salicornia bigelovii* plots, which were dominated by *S. bigelovii* seedlings. However, no species recruited well in the *S. virginica* plots. *S. virginica* seedlings swamped the available space, and *S. virginica* adults inhibited the recruitment of other species.

A balance should be struck between establishing cover, using one or more highly productive species, and establishing diversity by using the slower developing or rarer species. Species planting densities may be manipulated to bias initial community composition toward slow colonizers and away from the weedier species that will establish quickly from seed or asexual colonization. Such species can be locally segregated or introduced in smaller clusters to reduce their potential for crowding out other species. An important consideration in this regard is the proximity of the created site to source of propagules. Because species can establish quickly and in high density from seed imported from outside the system (Table 4.7), we recommend reducing the planting densities of species that are likely to invade quickly on their own (e.g., *S. virginica*). Such a strategy provides opportunities for the slower growing, less prolific species to become firmly established before others achieve exclusionary dominance.

4.4.3.2 Matching species to microhabitats

Elevation is a good general indicator of where to plant species, but it does not fully explain species distributions (Box 2.1). Elevation integrates several factors that influence species

Table 4.7 Frequency of occurrence (%) of naturally established species (seedlings) at three elevations (low = 0.85 m NGVD, mid = 1.0 m, and high = 1.3 m) at Crown Point, Mission Bay. Data represent the mean frequency of occurrence in ten 0.25-m^2 quadrats in each of 5 randomly located transects at each elevation. Species abbreviations are given in Table 4.4.

Elev	Sv	Sb	Se	Bm	Jc	Ml	Lc	Ds	Sm	Pi	Tc	Sf	Bh	Mn	As
high	58	22	30	22	20	14	4	8	8	6	0	0	4	2	2
mid	20	82	6	4	2	0	0	0	0	0	0	2	0	0	0
low	40	86	14	10	2	0	8	0	0	0	2	2	0	0	0
mean	39	63	17	12	8	5	4	3	3	2	1	1	1	1	1

performance, the most important of which is *hydrology* (Chapter 3). Hydrology represents water quality, salinity, hydroperiod, and circulation patterns. Hydrology is a function of elevation and other factors, including distance from shore, proximity to creeks and freshwater inflows, local topography, soil texture, and the biotic environment. While elevation affects the hydroperiod (depth and duration of tidal inundation), the distance from shore interacts with elevation to dampen tidal oscillations, with the lag of water moving over the landscape reducing both velocity and length of flooding. Topographic features such as berms or sinks determine local flooding and drainage patterns. Hummocks are exposed more rapidly and remain saturated for less time, while depressions retain water longer and may only drain during extended nonflood periods (Pollock et al. 1998). Soil drainage characteristics are a function of soil texture and organic matter content, which strongly interact with the underlying soil layers (e.g., a clay or sand lens) and location on the landscape (e.g., creek edge or center of plain). At a given elevation and landscape position, soils draining more rapidly will be less waterlogged and have higher redox potential. Marsh soils with high organic matter content will retain moisture longer than coarse sediments at lower elevations. Plant cover also interacts to slow tidal velocity and increase the deposition of sediments (Adam 1990).

The diversity of plant assemblages encountered in natural systems reflects the mosaic of small-scale heterogeneity (Bazzaz 1991). Coastal wetlands extend from high marsh to channel and bay margins (Chapter 2), with the intervening marsh plain cut by a network of tidal creeks (see Pestrong 1965). In southern California coastal wetlands, *Spartina foliosa* occurs along the edges of bays and large channels, while most of the salt marsh supports a variety of succulents. Although the marsh plain is relatively flat, this habitat is far from homogenous (Zedler et al. 1999). We recommend that salt marsh restoration projects incorporate as much of this heterogeneity as possible (Chapter 2). It is intuitive that species should grow best at their modal elevation. However, a species with a broad range might function differently at lower vs. higher elevation; that is, species roles may differ within microhabitats across the landscape. *Salicornia virginica* has a lanky, decumbent growth form within stands of *Spartina foliosa*, but it grows stiff, upright, and tall in well-drained sites, where its vertical structure provides perch sites for the endangered Belding's Savannah sparrow. We recommend that species be introduced most densely to preferred microsites near their modal elevations (determined from nearby reference marshes) and less densely over other portions of their full elevation range.

4.4.3.3 Planting density and clustering

Planting density should be maximized, while at the same time leaving open space for natural recruitment of any desired species that are likely to disperse into the restoration site. Cluster plantings accomplish both objectives by including dense plantings of poorly dispersed or rarer species and open spaces for volunteer recruitment. Facilitative or complementary interactions are known for salt marsh species in New England (Bertness and

Shumway 1993) and southern California (Callaway 1994), suggesting that seedlings or small plants can be introduced in clusters composed of species that naturally interact in similar microhabitats. We hypothesize that the diverse species clusters will colonize adjacent bare areas more quickly, although this strategy and the optimal composition of clusters needs further testing. Experiments with cluster plantings are underway at the 8-ha Model Marsh at Tijuana Estuary (Box 1.11).

Planting in higher density requires more plants — for example, planting on 33-cm centers requires 8× the number of plants as planting on 1-m centers. The cost of high-density planting can be reduced by planting smaller individuals, which are less expensive to grow and to plant (smaller holes and less digging). Planting high densities should accelerate ecosystem development, close the canopy sooner, and create cover and habitat for the animal community (Keer 1999). Low density plantings will establish cover in time, although it might take several seasons where soils are of low quality.

Bare space is desirable for natural recruitment (Table 4.7, Figure 4.12), but it can pose problems if propagules of native species are lacking when the site becomes available or if the soil dries and forms a salt crust that prevents recruitment. The bare space between cluster plantings would then be open to colonization by salt tolerant exotics such as *Parapholis incurva* in the high marsh (Table 4.2). Areas of bare space will increase with transplant mortality, and hypersalinity will develop rapdily in areas with little or no vegetation (Chapter 3), inducing further mortality (Figure 4.3). Thus, bare space can inhibit subsequent plant establishment (Bertness and Shumway 1993, Bertness and Hacker 1994). In southern California, bare, hypersaline patches are more likely to be colonized by vegetation in years of heavy winter rainfall, when seeds are being dispersed and when soils become saturated by rainfall during low tides (maximizing the leaching of salts and increasing germination rates). The task of restoration is to create such conditions on demand.

At the Tidal Linkage, we planted seedlings on 20-cm centers in 2 × 2-m plots with relatively low nutrient soil. With this approach, our multi-species plots had closed canopies (100% cover) after 18 months (Figure 4.13). Similar plantings on soil that was amended with composted kelp resulted in canopy closure after just 6 months (Figure 4.14).

4.4.3.4 Fertilizing transplants in the field

Once transplants have been introduced to the field, fertilizers can be used to increase plant size and hasten canopy development. Although herbivory can increase on tissues high in nutrient concentrations following fertilization (Section 7.4), our experience is that urea additions reduce insect herbivory (Boyer and Zedler 1996). Broadcasting dry fertilizer and rototilling it in prior to planting can enrich soil nutrients, but most of this will not benefit widely spaced plants. Excessive use of fertilizers can also cause local eutrophication and nuisance blooms of macroalgae (Section 7.5). The high soil nutrient levels associated with broad fertilizer application may also encourage the invasion of weedy exotic species over the desired native introductions. Plant species differ in their nitrogen (N) demand and N-use efficiency, and fertilizing mixed-species stands can lead to changes in composition (Boyer and Zedler 1999).

On the other hand, marshes with mineral soils or dredge spoils are notoriously low in nutrients (Chapter 3), and fertilization can help transplants develop the biomass needed to buffer them against environmental stress (Chapter 7). How long the effects of fertilization will extend into subsequent growing seasons is questionable. Broome (1990) suggested long-term effects for North Carolina wetlands, but sandy marshes planted with *Spartina foliosa* in San Diego Bay grew better above and below ground, but only during the growing seasion when fertilizers were applied (Gibson et al. 1994, Boyer and Zedler 1999, PERL unpublished data). This one-season effect was found for multiple applications within a

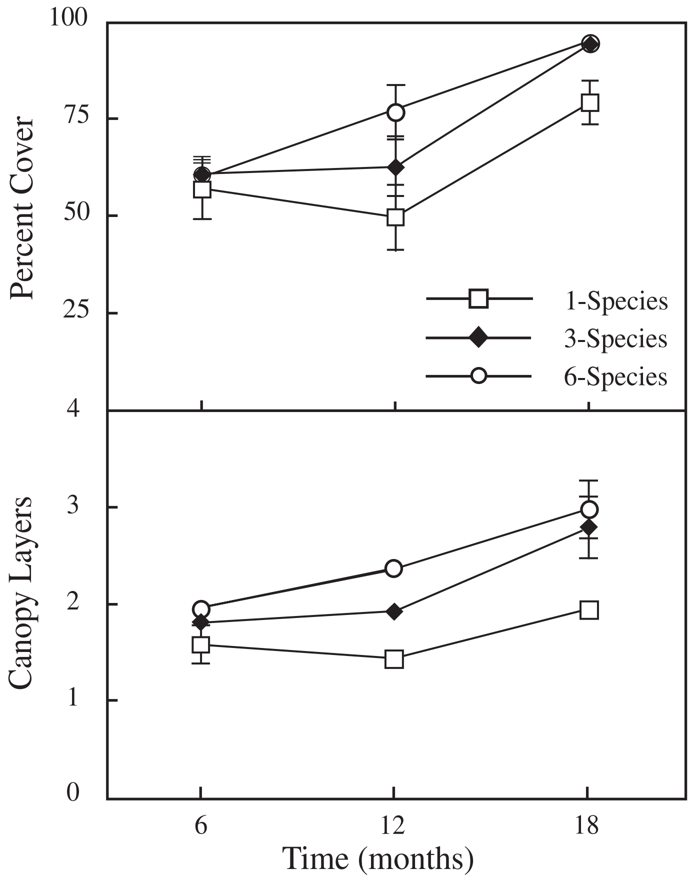

Figure 4.13 The development of cover and canopy layering for 1-, 3-, and 6-species assemblages at the Tidal Linkage after 6, 12, and 18 months. Cover was assessed by line intercept. Layering is the number of canopy hits per pin using point intercept. Data from Keer 1999, The effects of species richness and composition on salt marsh canopy architecture. Master's thesis. San Diego State University, San Diego, CA.

year, as well as multiple applications over multiple years; when fertilization ceased after 1 to 5 years, plant growth was consistently impaired in the subsequent year (PERL unpublished data). Our studies indicate that there are clear benefits to using fertilizers and soil amendments, but more work is underway to compare soil amendments for different marsh communities.

We recommend that any fertilizers chosen be carefully applied. One effective and widely used method of fertilizing individual transplants is to place slow-release fertilizers (e.g., Osmocote™) in the transplanting hole before adding the plant. Up to 30 g of fertilizer (e.g., N:P:K = 14:14:14) works well in most cases (see Broome 1990). Soluble fertilizers can also be used, although the nutrients are not available for as long a period and reapplications will be needed (Shisler 1990). Solutions should be placed in holes 5 to 10 cm away from the transplant roots, depending upon plant strength and soil characteristics (Broome et al. 1983).

4.4.3.5 Irrigation during establishment

Watering is critical to the initial survivorship of many species, especially if the plants are not exposed to regular tidal flooding (Chapter 7). Plants at higher elevations or those

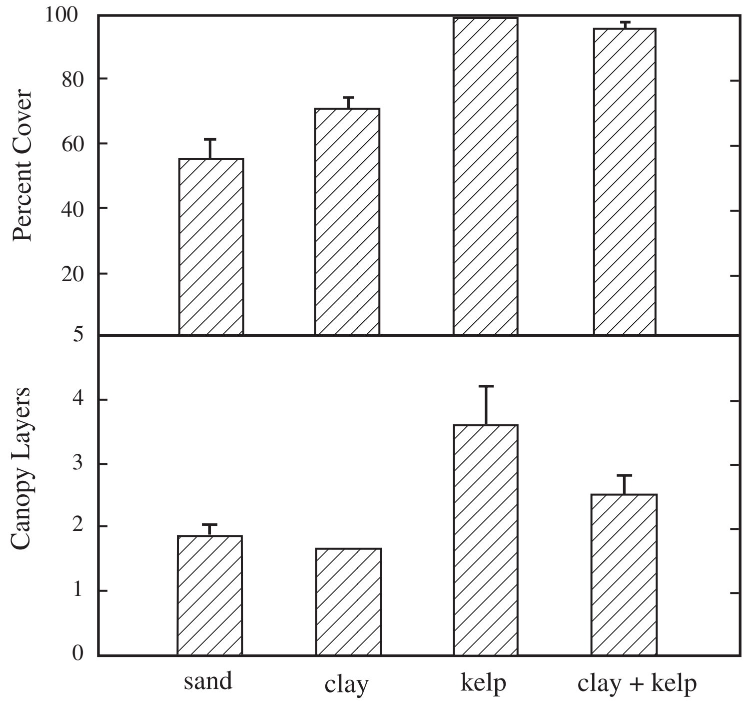

Figure 4.14 The development of cover and canopy layering for 8-species assemblages in soil treatment plots at the Tidal Linkage after 6 months. Sand was unamended substrate at the restoration site. Clay sediment was salvaged from a channel cut through a mudflat. Kelp was 1/3 kelp amendment + 2/3 *in situ* sediment (vol:vol). Clay+kelp was 1/3 kelp amendment + 2/3 clay sediment (vol:vol). Cover was assessed by line intercept. Layering is the number of hits per brass rod dropped through the canopy.

planted during a neap tide series — a convenient time for working in intertidal marshes — may die before their roots can grow to the water table. Smaller plants may be more susceptible to desiccation stress, with higher surface:volume ratios, fewer belowground reserves, and a shallower root system, but no plant will survive prolonged drought. As soils dry, hypersalinity adds to the stress of desiccation. Periodic watering during the establishment phase will reduce salinity and help plants develop deep root systems. Despite the critical importance of early watering, irrigation should not be maintained beyond the establishment phase. Extended irrigation benefits growth, but it also encourages the establishment of exotic plants, especially in the higher elevations. Many exotic species establish rapidly with freshwater influence, and, once established, they are difficult to remove (Chapter 7, Callaway and Zedler 1998). Consequently, overwatering should be avoided and irrigation terminated as soon as practical. Soils should also be allowed to dry to a certain extent between waterings to discourage extensive surface root growth. Soils that are intermittently drained encourage deeper root growth and ensure that plants survive once irrigation is suspended.

4.5 Genetic considerations

4.5.1 Genetic diversity

Genetic diversity is a measure of the number and frequency of alleles — the various forms of each gene in the pool of genes shared by a population. Theory predicts that genetic

variation is critical for populations to adapt to changing or heterogeneous environments (Wright 1978, Hartl and Clark 1989), such as those characterized by urban development, spatial fragmentation, pollution, natural climatic variation, or cycles of pathogen and herbivore impacts (Silander 1984, Burdon 1985, Nevo et al. 1986, Huenneke 1991). Consequently, the level of genetic diversity indicates the long-term potential of a population to survive (Van Treuren et al. 1991). Although maintaining genetic diversity has become a guiding principle in conservation biology, it is usually overlooked in planning restoration. The lack of sufficient genetic diversity might reduce the persistence of populations planted into dynamic environments (Williams and Davis 1996). We currently know little of the fate of small populations established with low genetic diversity, especially among rare or endangered species (Helenurm and Parsons 1997). However, Denise Seliskar and Jack Gallagher are showing that genetic background plays a major role in population performance and whole-ecosystem functioning in a Delaware restoration site (Seliskar 1995). Clones of *Spartina alterniflora* from Maine, Delaware, and Georgia show genetically determined differences in structure that influence nearly every salt marsh attribute, including productivity and standing crop, decomposition rate, invertebrate use, and fish abundance (ibid. and J. Gallagher, *personal communication*).

Restoration sites offer excellent opportunities to study the long-term effects of genetic diversity on the survivorship of founder populations (Montalvo et al. 1997, van Andel 1998). In this section, we discuss how the level of genetic diversity may affect restored populations. Specifically, we discuss:

- how the genetic structure of donor populations determines the potential genetic diversity of restored populations;
- the benefits of using locally adapted plants in restored populations;
- the dangers of genetic pollution (foreign genes invading the gene pool); and
- how to increase the genetic diversity of locally adapted plants at a restoration site.

4.5.1.1 Genetic structure

Genetic variation is a measure of the diversity of alleles, and genetic structure is a measure of how these alleles and genotypes are spatially distributed across the landscape. Highly structured populations are characterized by the nonrandom distribution of alleles both within and among subpopulations or populations. That is, the diversity of alleles is not homogeneously distributed across hierarchical scales. Variation within populations means that individuals carry different alleles, suggesting a relatively high degree of heterozygosity — an indication of the potential for the population to adapt to changing conditions (Futuyma 1998). Variation among populations or subpopulations means that groups have diverged with respect to allelic frequencies and/or identities. Divergence occurs among relatively isolated groups with little gene flow, coupled with genetic drift or divergent selection pressures. Consequently, genetic differentiation may reflect important adaptations to local environmental conditions (Walley et al. 1974).

Where the level of genetic diversity in a newly created population is a concern, it is important to know the genetic structure of donor populations (McCue et al. 1996). If the pool of alleles is spread evenly across a population, i.e., if the existing diversity is randomly distributed among all individuals, then relatively few donors may encompass all of the genetic diversity available. However, if the donor population is highly structured, then donors must be sampled widely to maximize the resulting diversity within a created population. Establishing genetically diverse populations in a new site is doubly important because small founder populations can cause a genetic bottleneck, with the initial members determining the adaptive potential for generations to come. If one does not know the genetic composition of the donor population, the appropriate strategy is to sample broadly

within nearby source populations. We recommend this as a routine approach for propagule collection.

4.5.1.2 Local adaptation and genetic pollution

Although most species show plasticity in response to a range of environmental conditions as an adaptive trait (Bradshaw 1965, Bazzaz 1996), a wealth of studies indicate that plants are genetically adapted to the local biotic and abiotic environment (Turkington and Harper 1979, Gray 1987, Bradshaw and Hardwick 1989). The use of ecologically relevant genotypes in restoration should improve the development of viable plant communities. Conversely, using plants adapted to foreign conditions could impair long-term performance. The issue becomes more complex when one considers that initial features of a restoration site, such as the soil texture and organic matter content, might not resemble conditions at reference or donor sites. Still, a community of locally adapted populations should respond well to factors such as weather patterns, hydrology, herbivores, pollinators, and pathogens. Consequently, there is increasing interest in matching locally adapted genotypes to restoration sites and to specific microsites within them (Fenster and Dudash 1994, Guerrant 1996, Montalvo et al. 1997).

The danger in using plants that are not locally adapted may not be readily apparent. Nonlocal genotypes can perform well initially and come to dominate the developing community. But plants with high establishment potential might not be adapted to withstand recurrent or episodic events, such as flooding, drought, pathogen attack, or various human impacts. Genotypes that initially dominate restoration sites could succumb to selection after the first few years of growth.

Delayed mortality is just one price of importing nonlocal plants. "Genetic pollution" occurs when foreign genotypes hybridize with local stock, so that poorly adapted alleles invade the local gene pool (Montalvo et al. 1997). In good years, first-generation hybrids could grow exceptionally, spreading foreign alleles to nearby natural populations (at least for species that commonly reproduce from seed). Thereafter, poorly adapted alleles could reduce the fitness of the restored populations and affected natural populations. This may be doubly harmful if recombination serves to break up coadapted traits — closely associated alleles that interact positively, but not when combined with alleles from other populations. Fitness of subsequent generations can also be reduced by outbreeding depression (Guerrant 1996, Montalvo et al. 1997). While these concerns are reduced for species that rely on vegetative reproduction, the potential for problems is easily avoided by using locally adapted and genetically diverse stock rather than relying on unknown nursery material or commercial seed sources (Section 4.2.3).

Although the importance of establishing genetically diverse populations is generally recognized in biological conservation (Lande 1988), demographic factors such as life history and dispersal ability dominate the planning of most introductions. Moreover, few projects have the resources needed to address the genetic structure of source populations. In the absence of this knowledge, we recommend that donor populations be sampled widely to reflect and take advantage of the breadth of available genetic diversity. If remnant populations are available at the restoration site, this preadapted stock should be salvaged and replanted. Species that are abundant over a broad area might have relatively high rates of gene flow, but ecotypic differentiation may still promote local genetic variation that will be valuable at the restoration site. We suggest matching morphotypes with microhabitats similar to those from which they were collected in order to take advantage of the existing genetic diversity.

Because founder populations can determine the long-term prospects of restored populations, we strongly recommend that restorationists:

- use locally adapted material rather than rely on nursery or commercial sources of unknown origin;
- collect widely from donor populations to sample the breadth of available genetic diversity;
- sample from remnant populations or communities similar to targets in proximity to the restoration site; and
- make use of ecotypic variation, matching morphotypes to the type of microhabitat from which they came.

4.6 Summary

We recommend:

- salvaging plants, seed banks, and soils from construction sites, recognizing that these represent the site's historical composition and gene pools;
- combining the designed and self-designed approaches, planting the species that need to be introduced, and allowing good colonizers to establish on their own;
- using seeds, seedlings, or cuttings to introduce native species;
- collecting propagules widely within local sites to increase genetic diversity within the population introduced;
- propagating and growing material with care, to ensure that healthy plants are used;
- timing the planting to take advantage of higher soil moisture and lower soil salinity;
- matching species to their preferred microhabitats (e.g., modal elevation) to reduce environmental stress, recognizing that a 10 to 20-cm error in vertical position can be fatal; and
- introducing the maximum diversity of native species.

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Plates

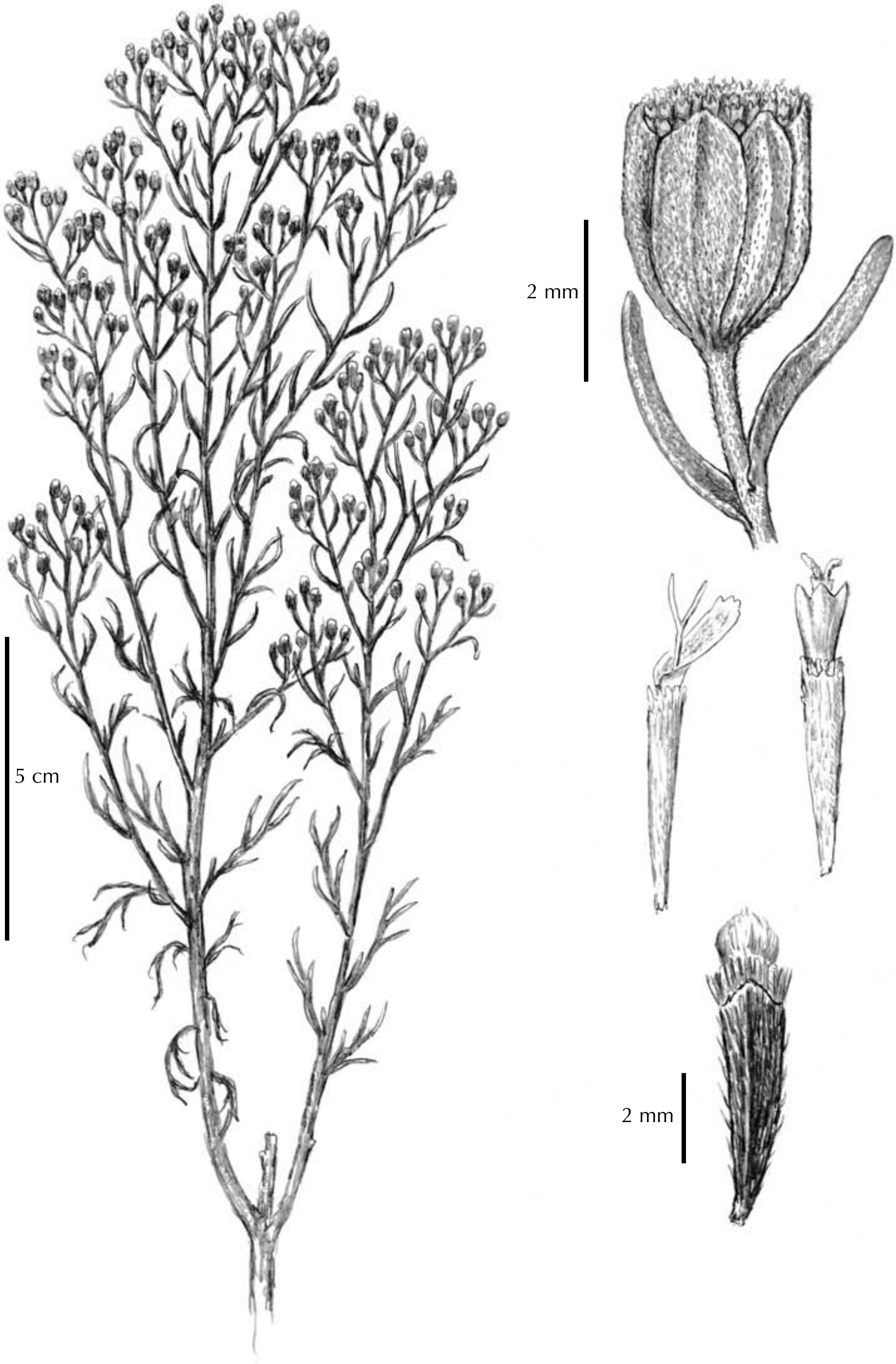


Plate 1. *Amblyopappus pusillus*. McIntire Drawings, © 1999 by Zedler.

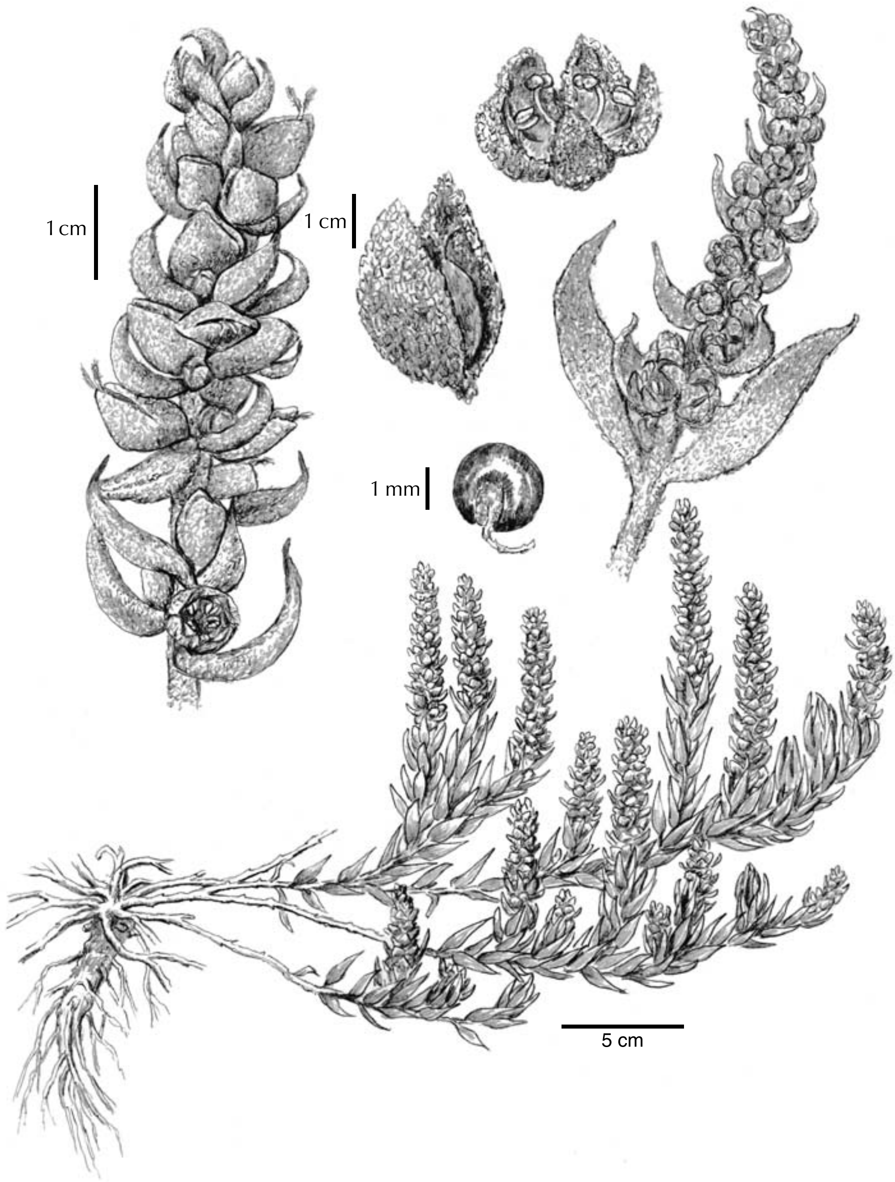


Plate 2. *Atriplex californica*. McIntire Drawings, © 1999 by Zedler.

Plate 3. *Atriplex triangularis*. McIntire Drawings, © 1999 by Zedler.

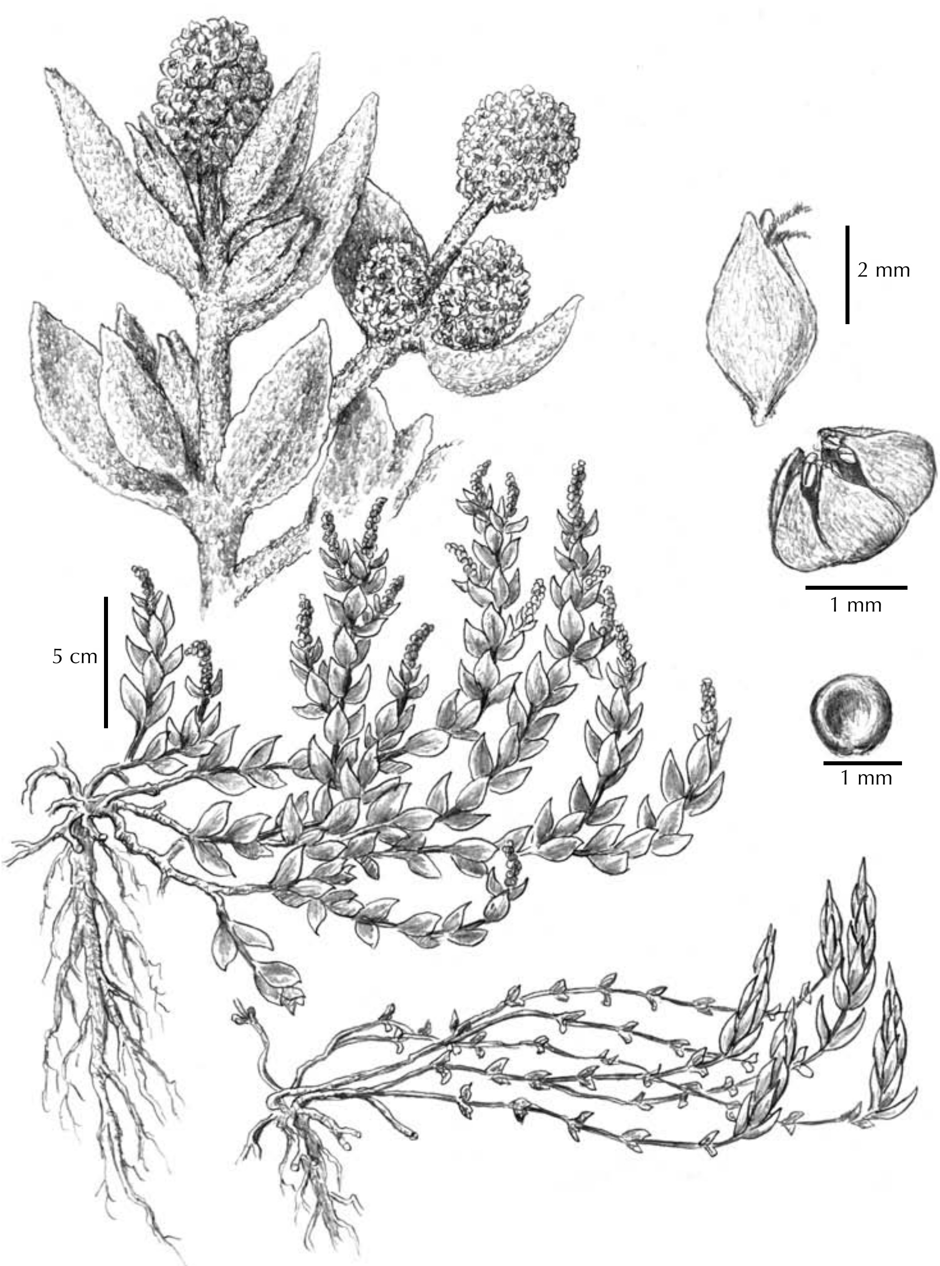


Plate 4. *Atriplex watsonii*. McIntire Drawings, © 1999 by Zedler.

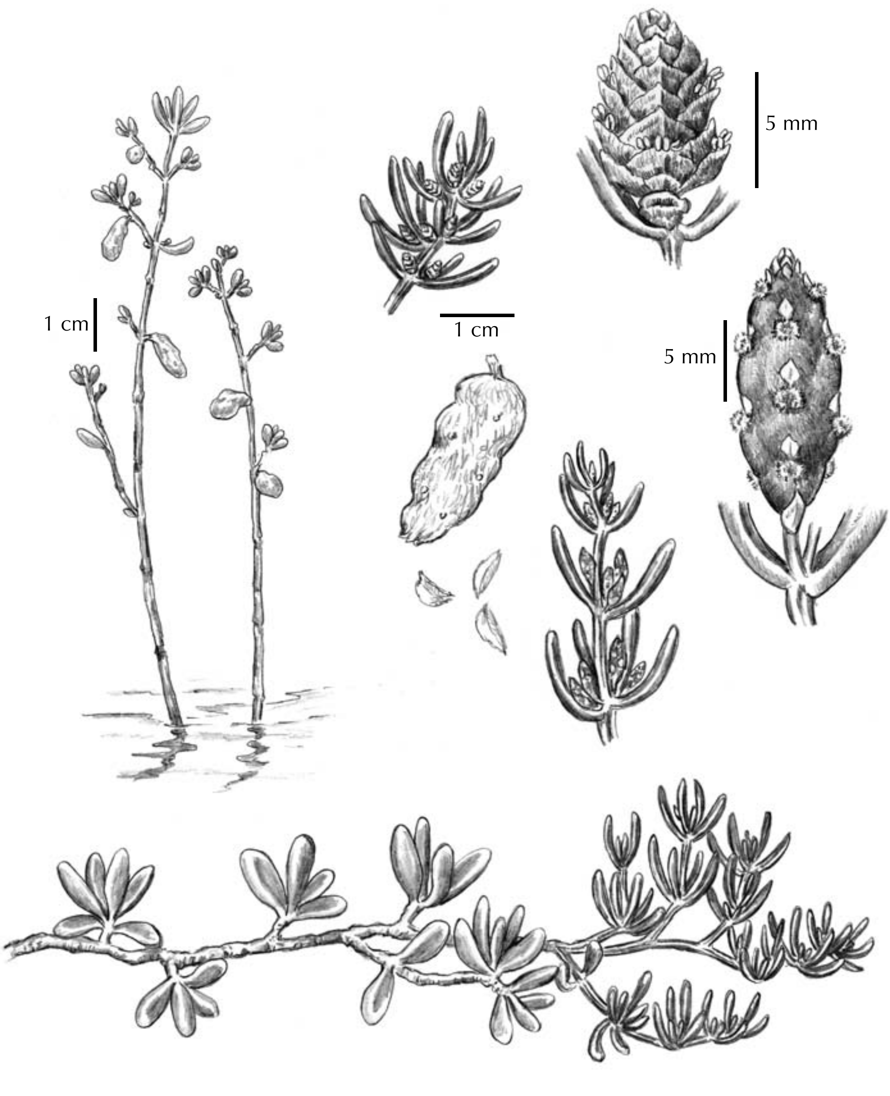


Plate 5. *Batis maritima*. McIntire Drawings, © 1999 by Zedler.

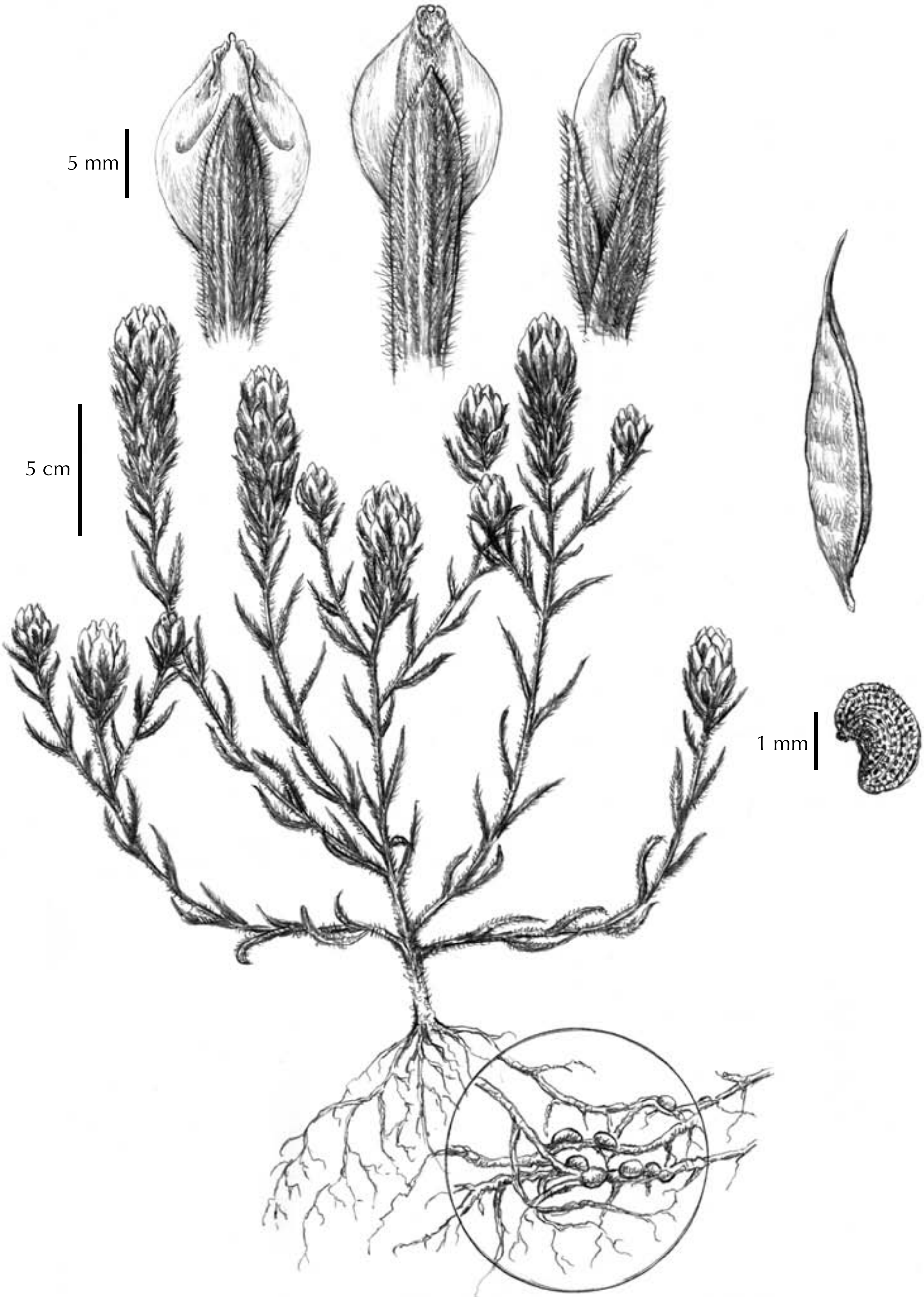


Plate 6. *Cordylanthus maritimus* ssp. *maritimus*. McIntire Drawings, © 1999 by Zedler.

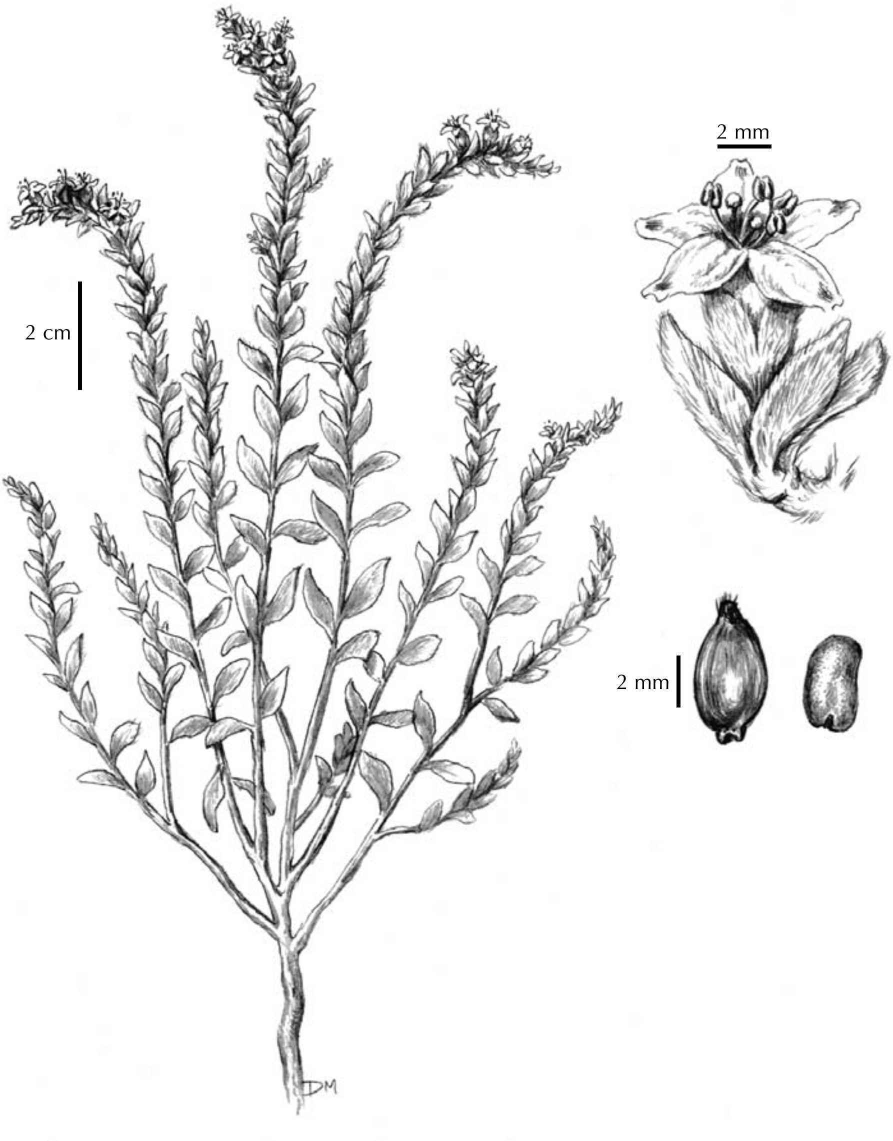


Plate 7. *Cressa truxillensis*. McIntire Drawings, © 1999 by Zedler.

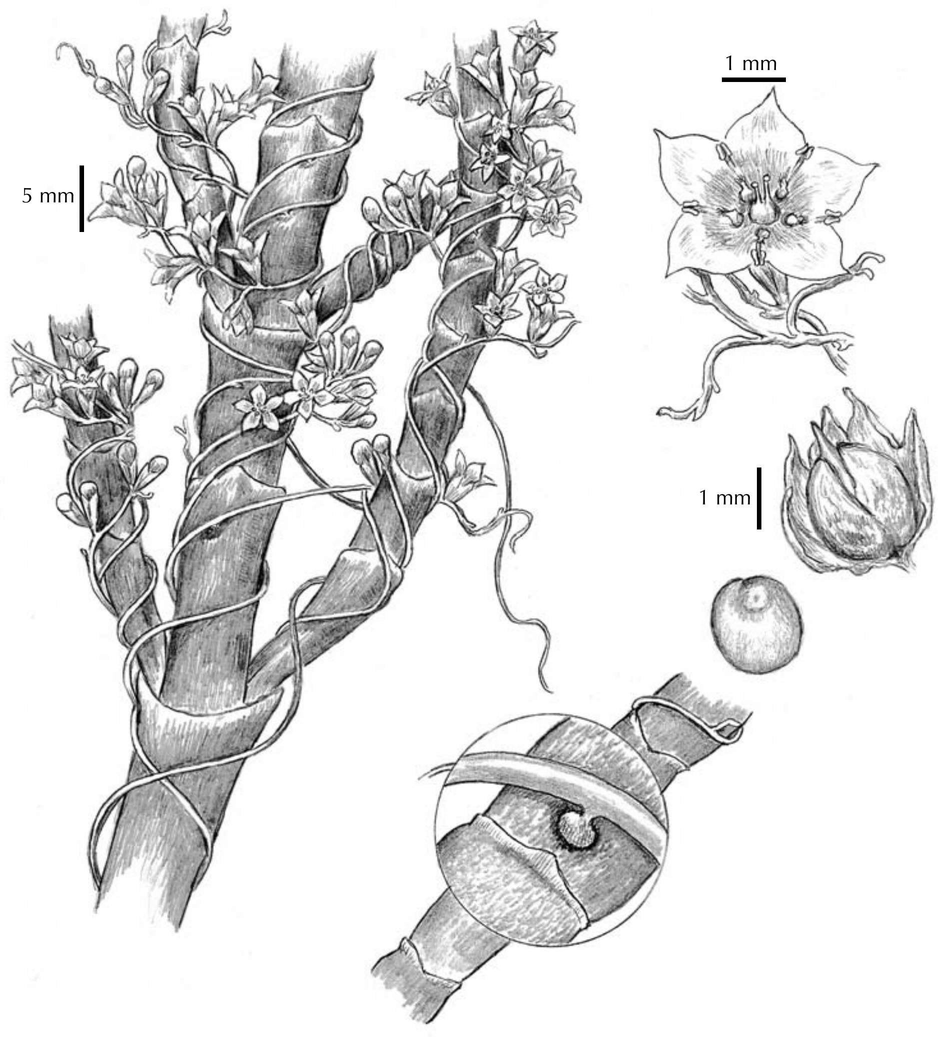


Plate 8. *Cuscuta salina*. McIntire Drawings, © 1999 by Zedler.

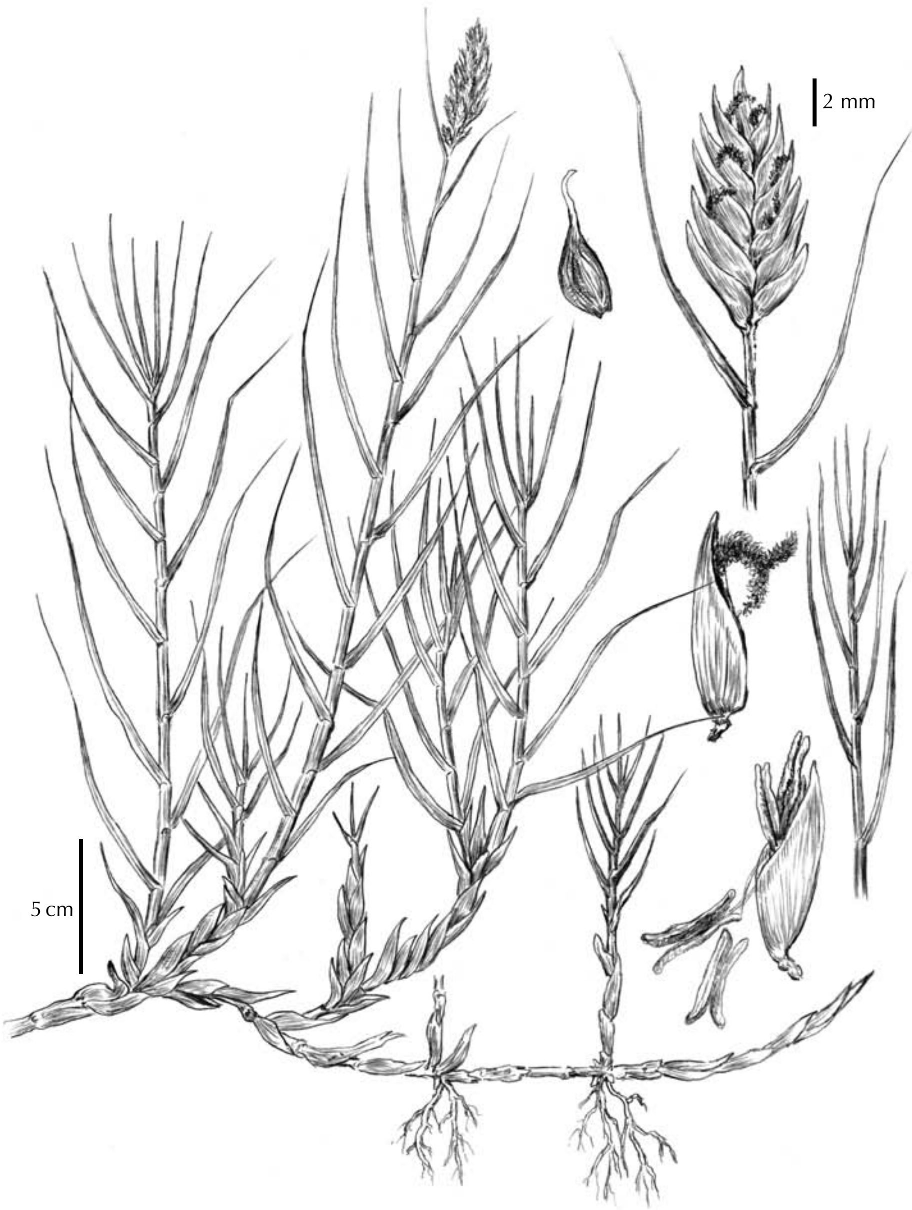


Plate 9. *Distichlis spicata*. McIntire Drawings, © 1999 by Zedler.

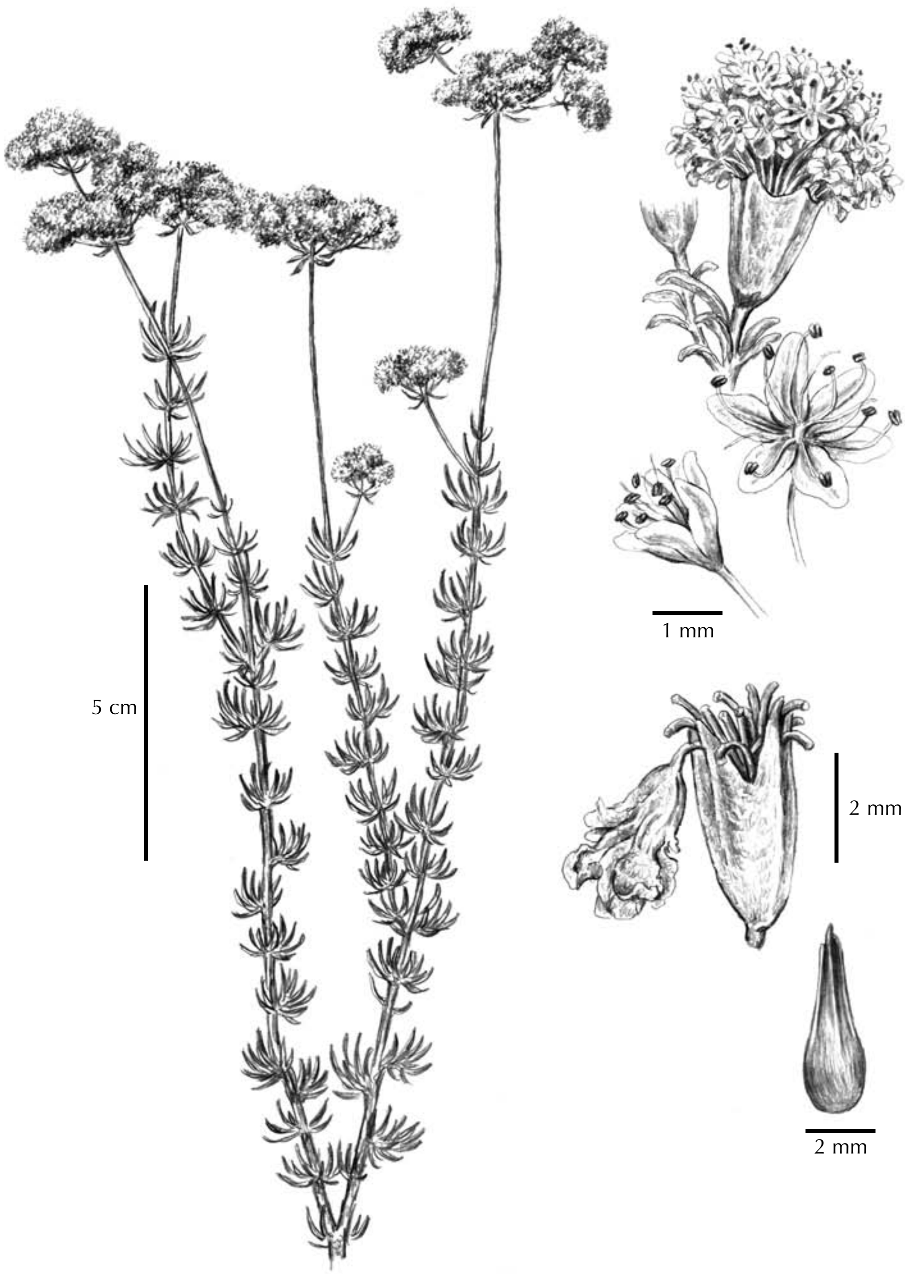


Plate 10. *Eriogonum fasciculatum*. McIntire Drawings, © 1999 by Zedler.

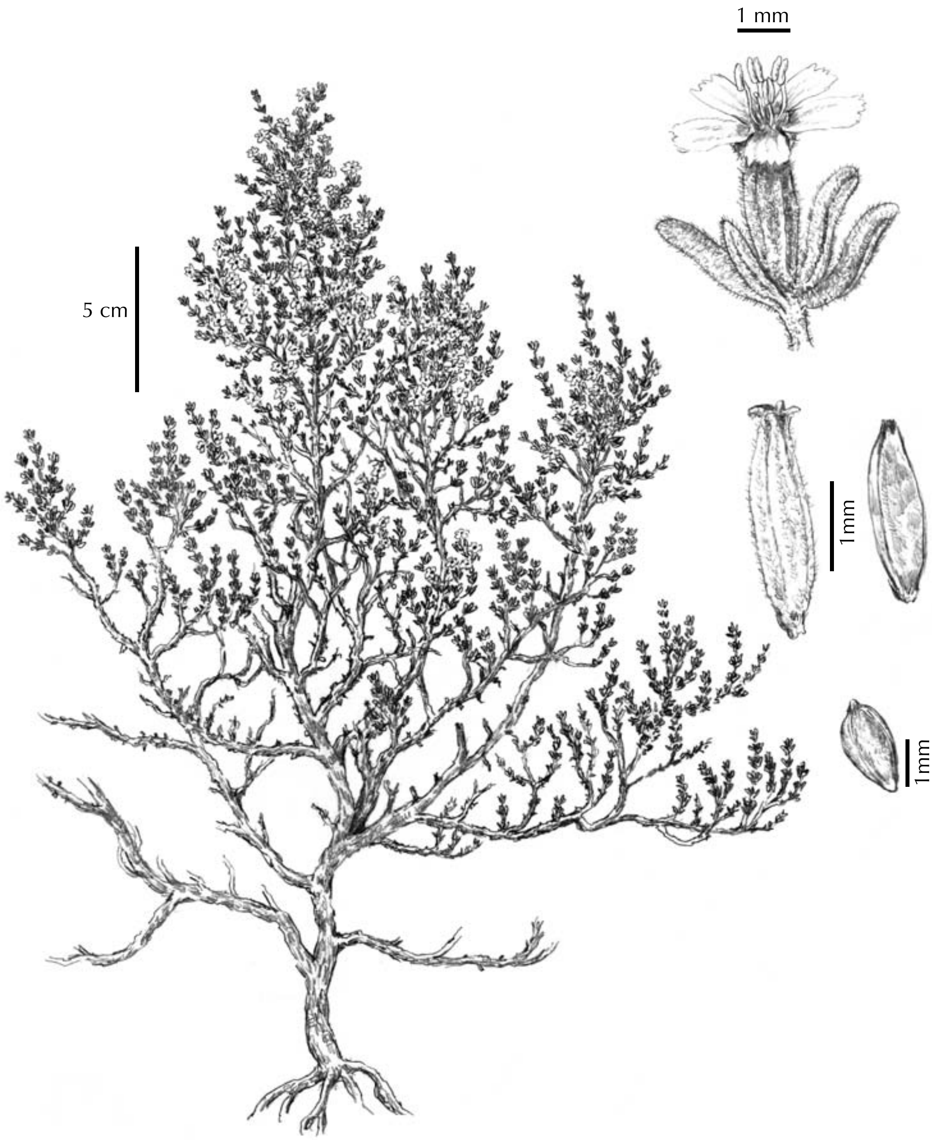


Plate 11. *Frankenia palmerii.* McIntire Drawings, © 1999 by Zedler.

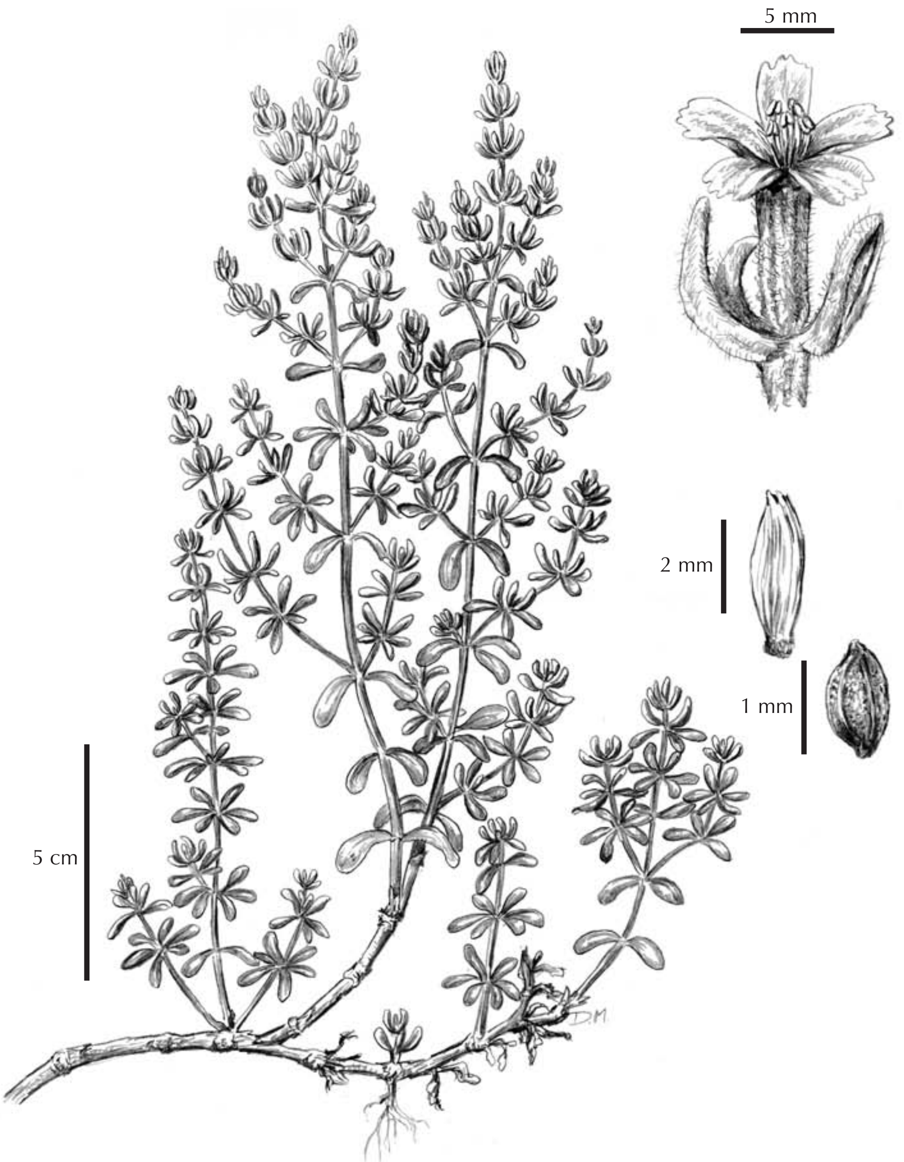


Plate 12. *Frankenia salina*. McIntire Drawings, © 1999 by Zedler.

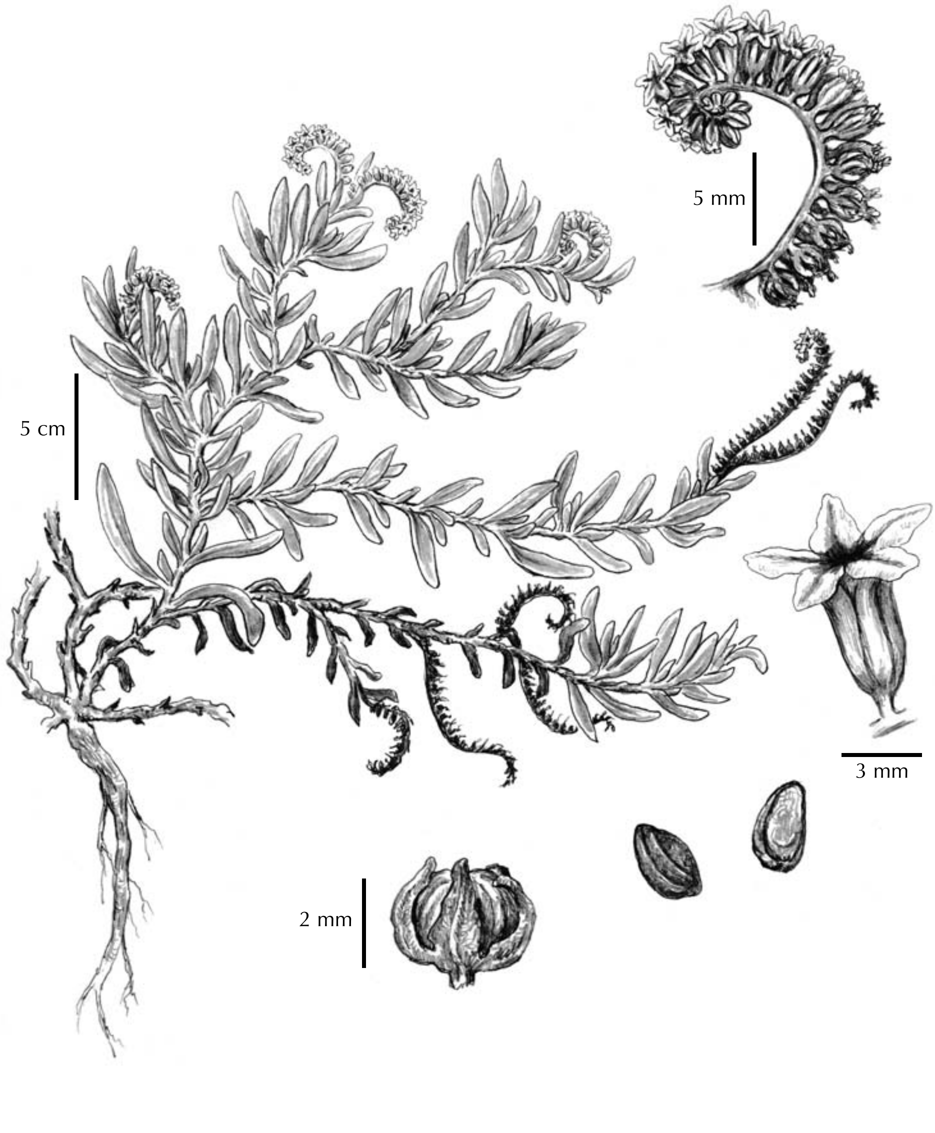


Plate 13. *Heliotropium curassavicum*. McIntire Drawings, © 1999 by Zedler.

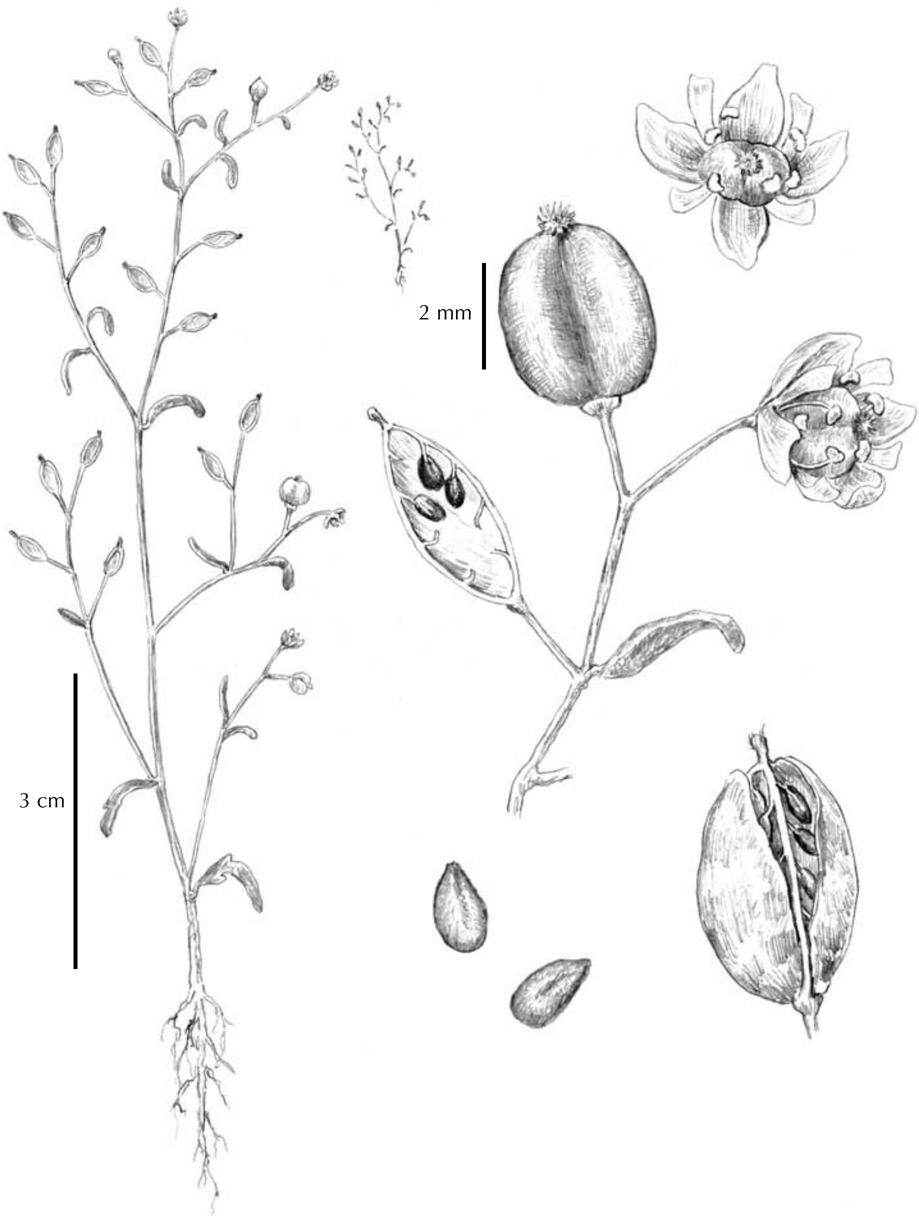


Plate 14. *Hutchinsia procumbens*. McIntire Drawings, © 1999 by Zedler.

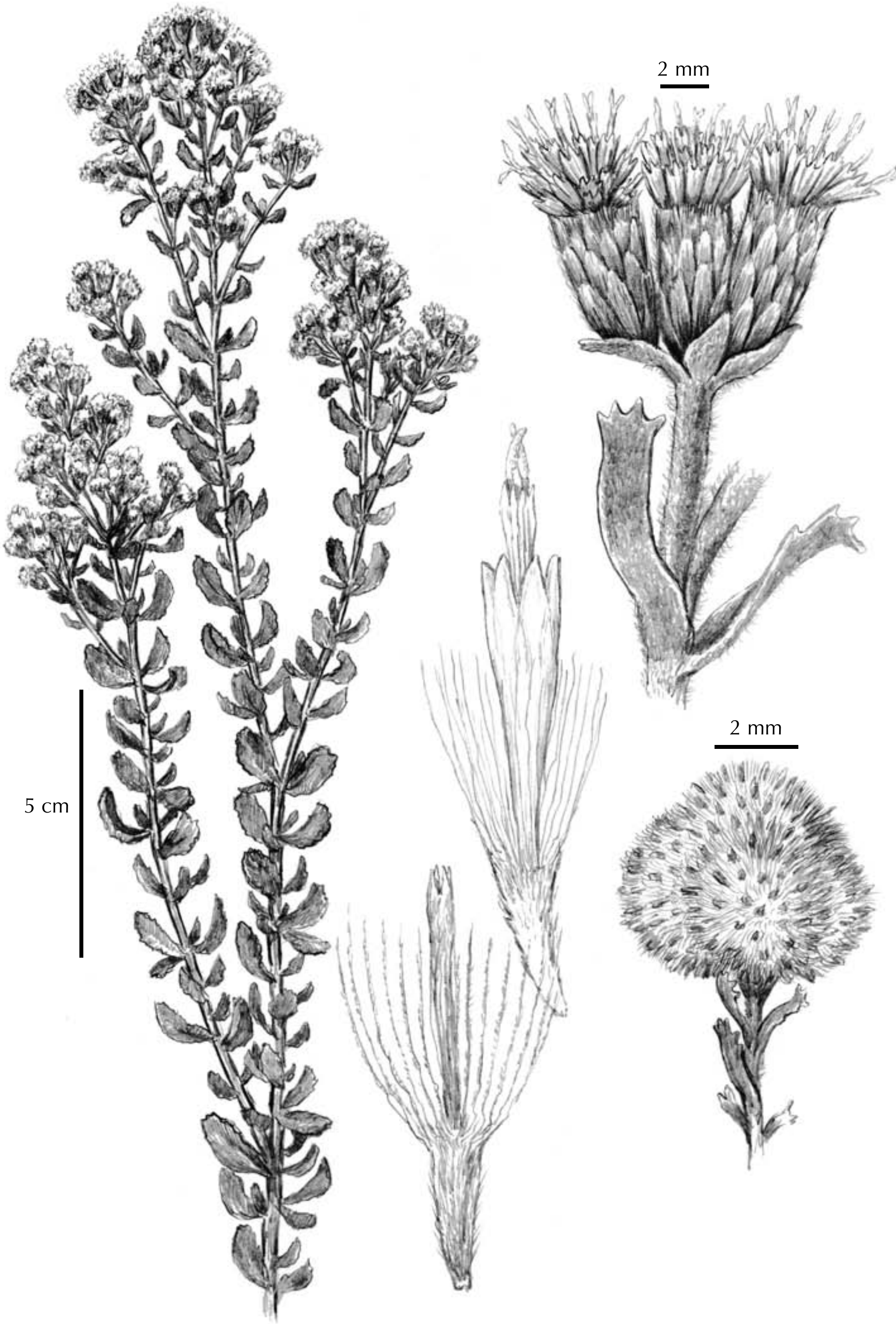
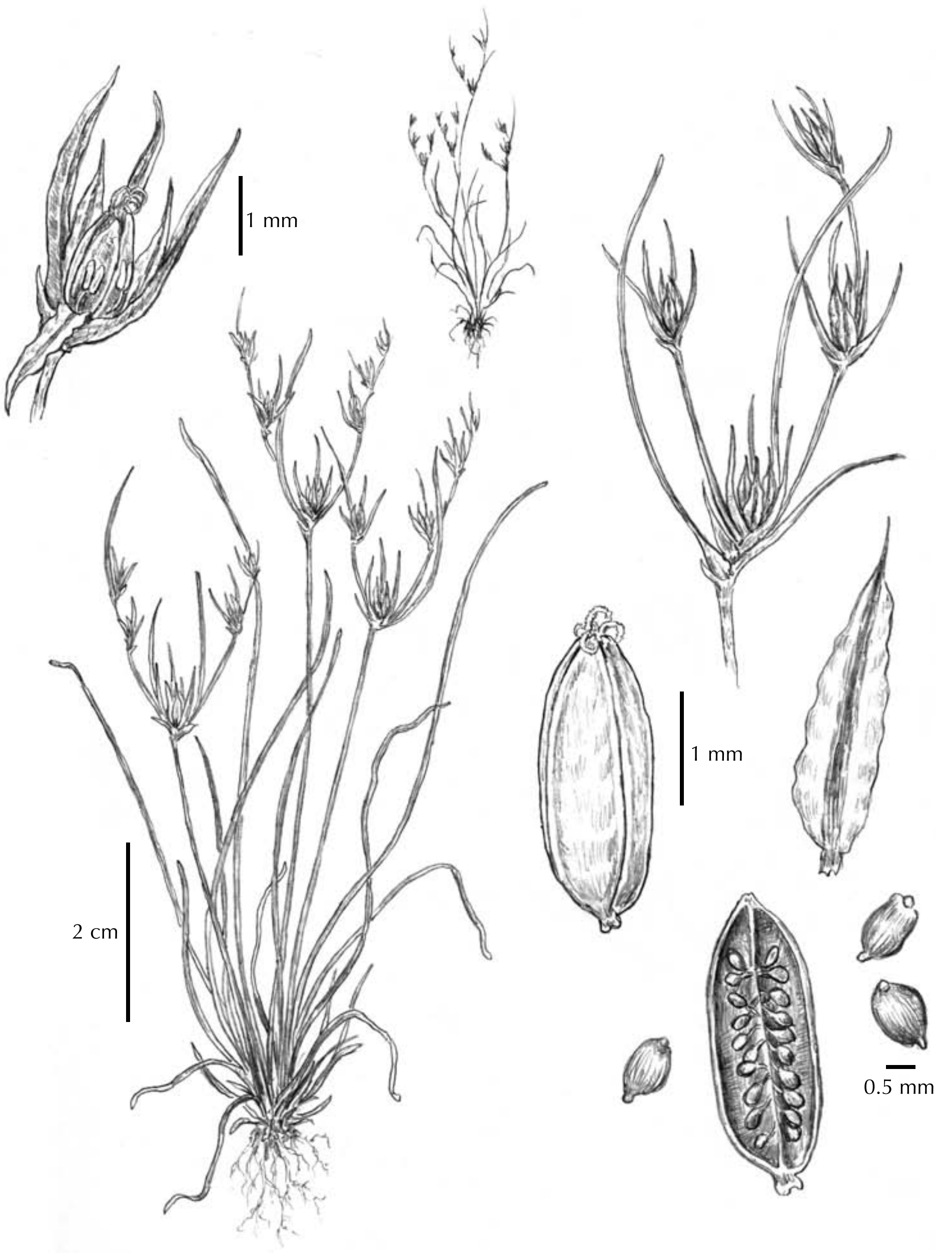


Plate 15. *Isocoma menziesii* var. *vernonioides*. McIntire Drawings, © 1999 by Zedler.

Plate 16. *Jaumea carnosa*. McIntire Drawings, © 1999 by Zedler.

Plate 17. *Juncus acutus*. McIntire Drawings, © 1999 by Zedler.



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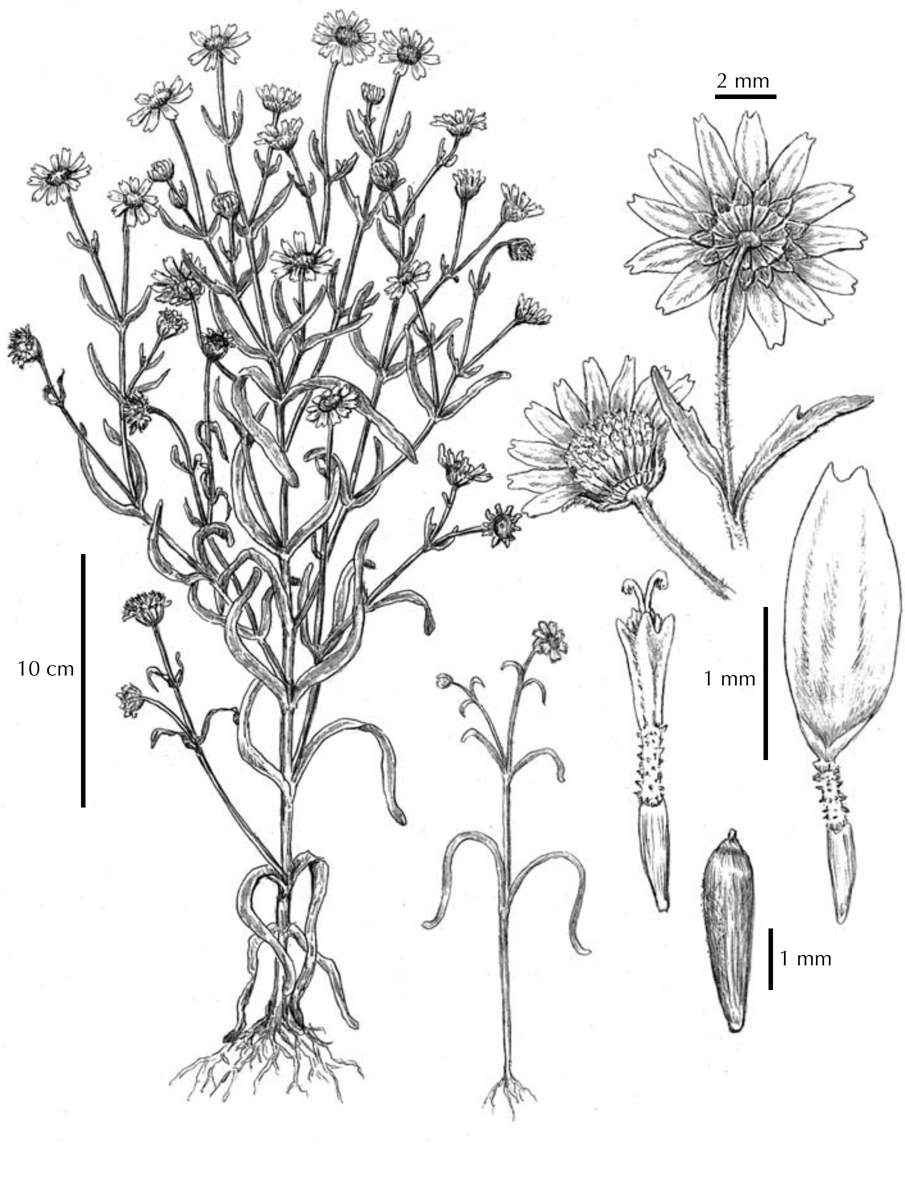

Plate 19. *Lasthenia glabrata* ssp. *coulteri*. McIntire Drawings, © 1999 by Zedler.

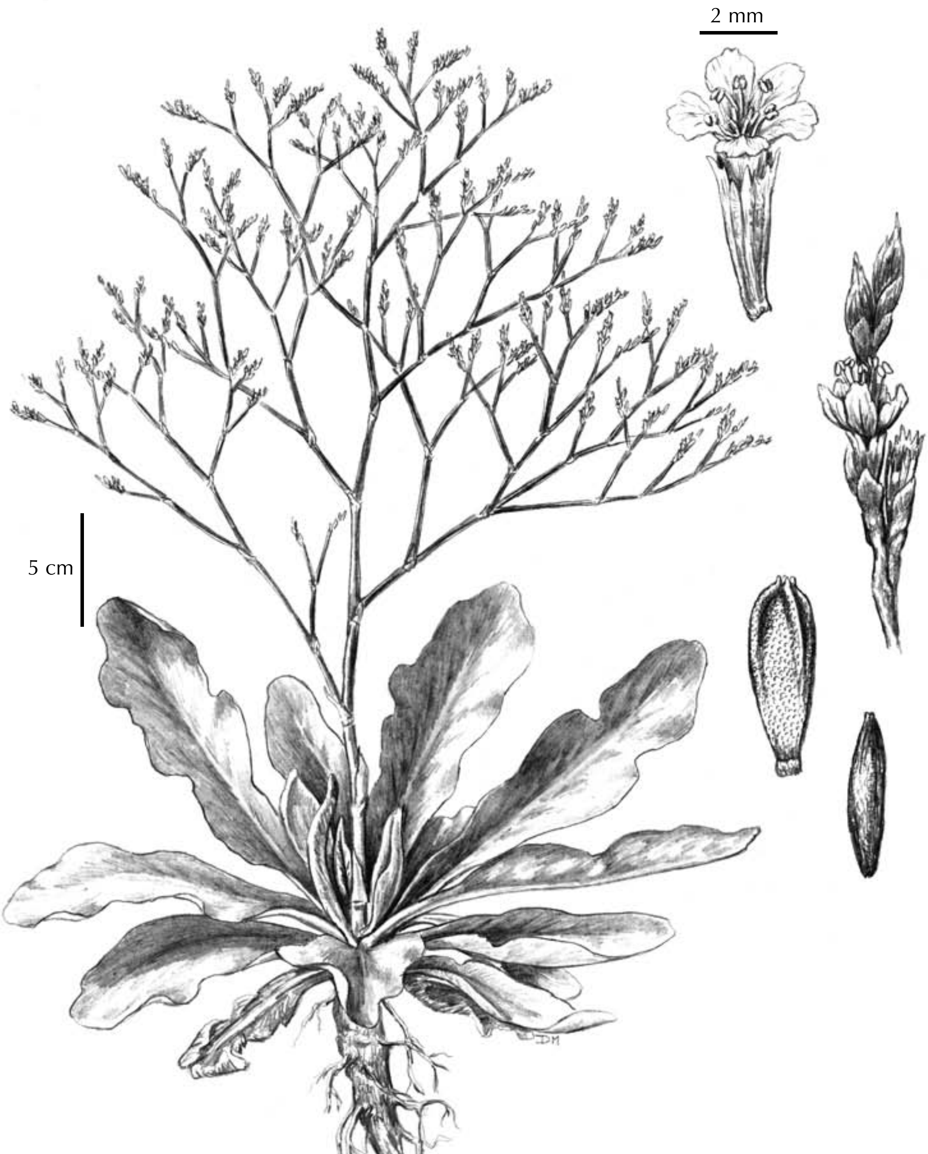

Plate 20. *Limonium caliofornicum*. McIntire Drawings, © 1999 by Zedler.

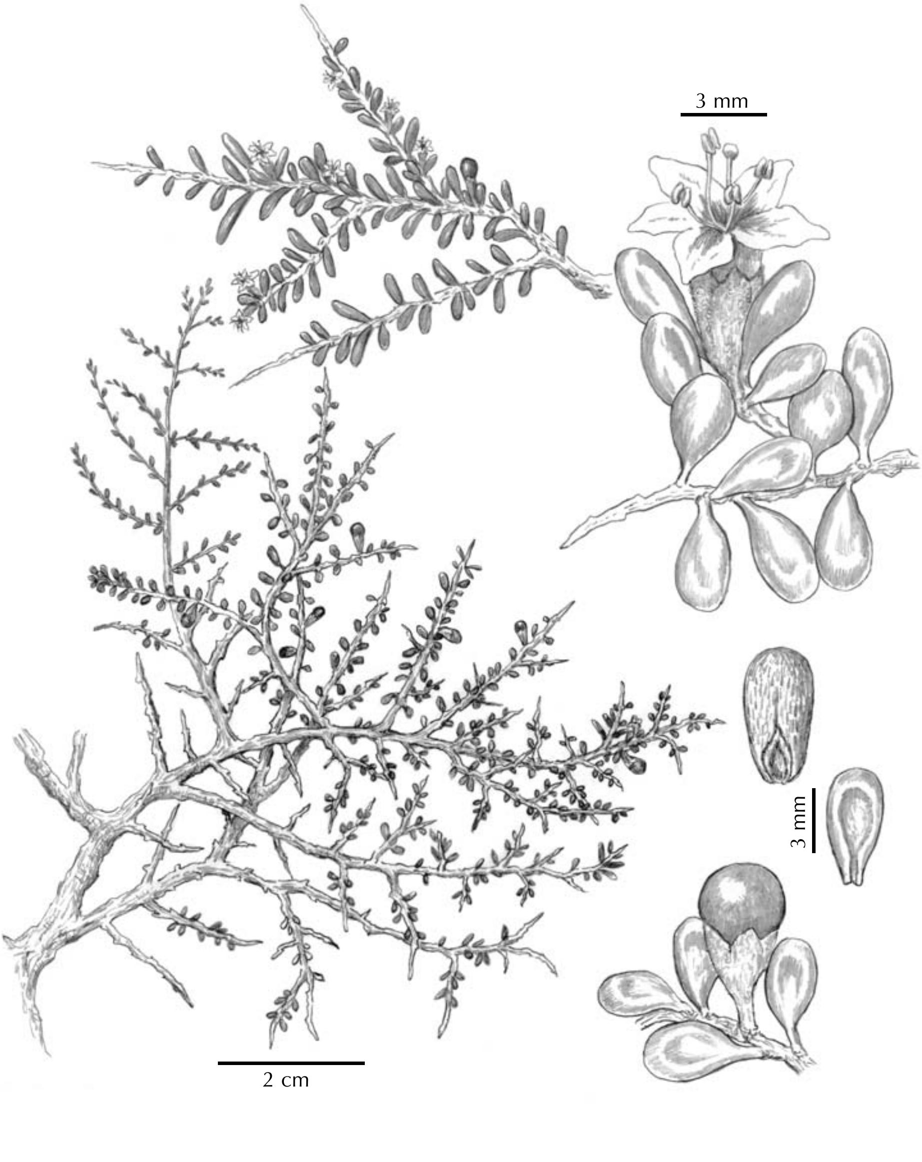

Plate 21. *Lycium californicum*. McIntire Drawings, © 1999 by Zedler.

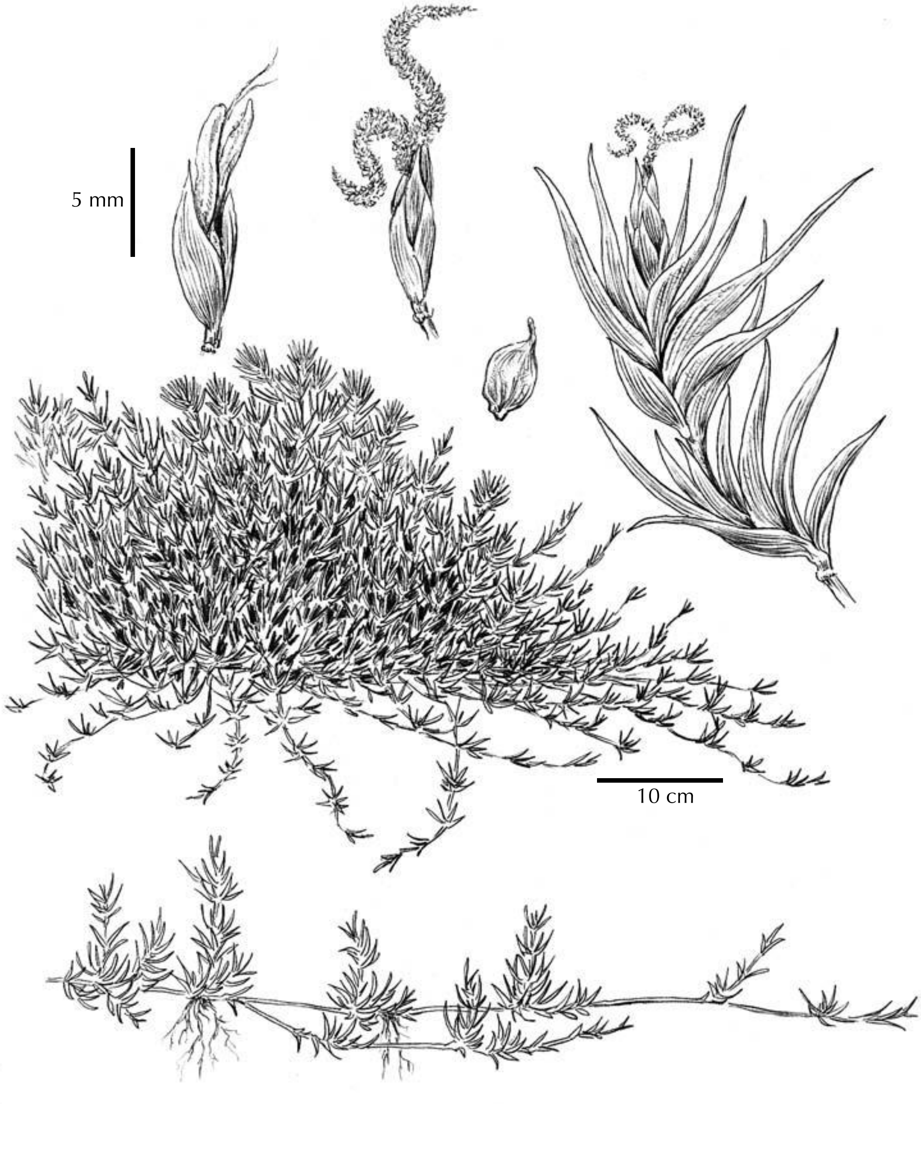

Plate 22. *Monanthochloe littoralis*. McIntire Drawings, © 1999 by Zedler.

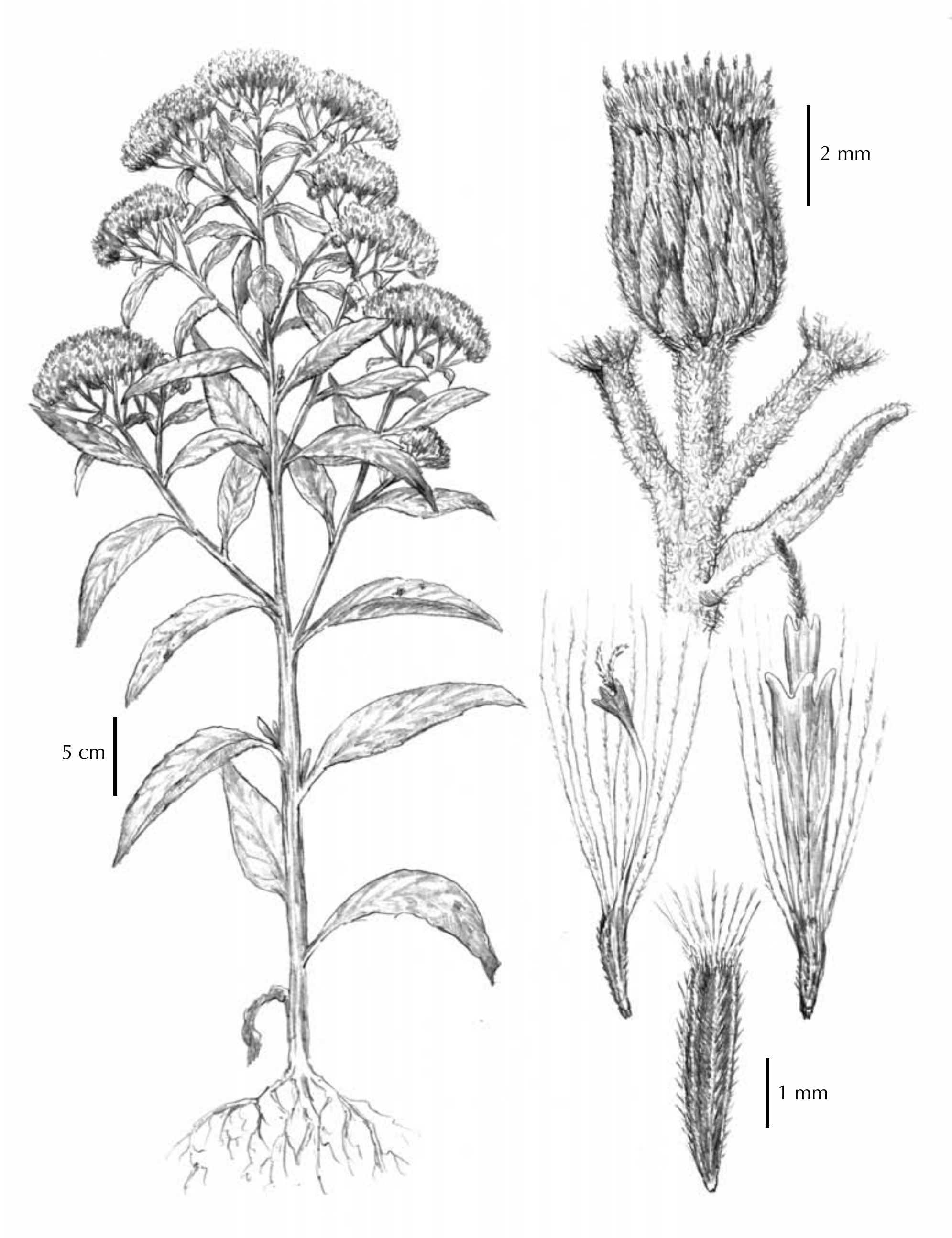

Plate 23. *Pluchea odorata*. McIntire Drawings, © 1999 by Zedler.

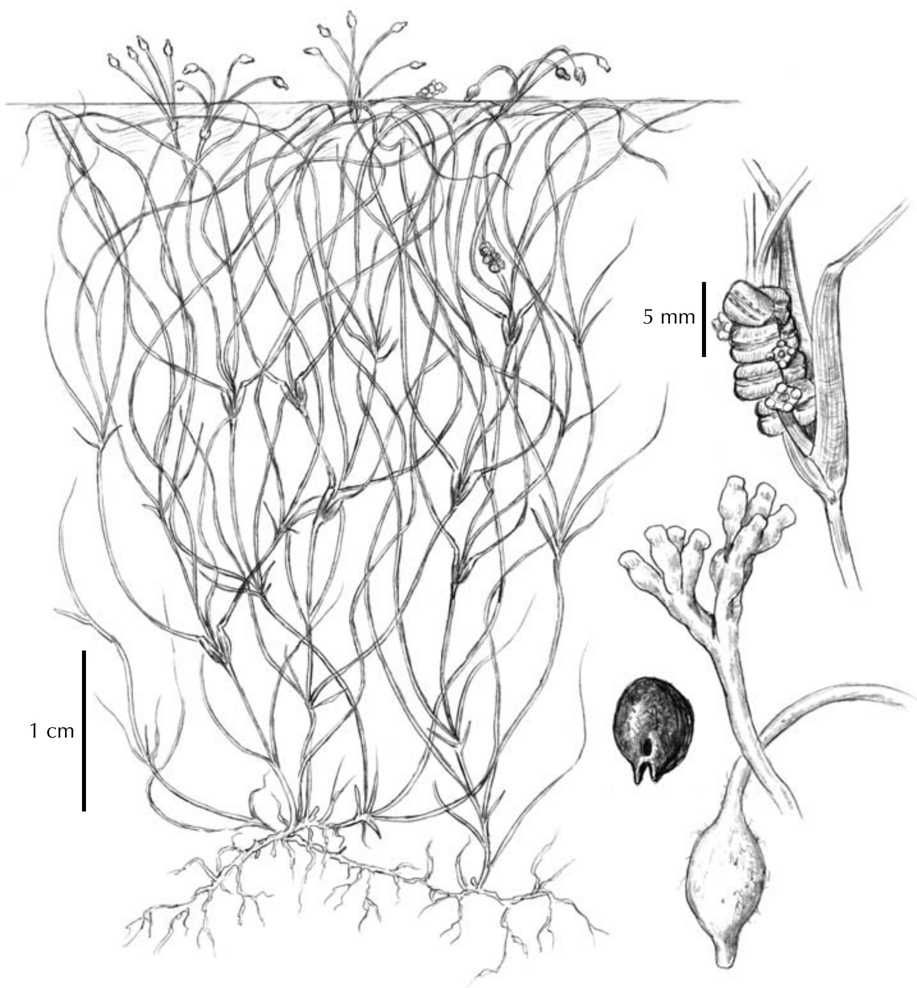

Plate 24. *Ruppia maritima*. McIntire Drawings, © 1999 by Zedler.

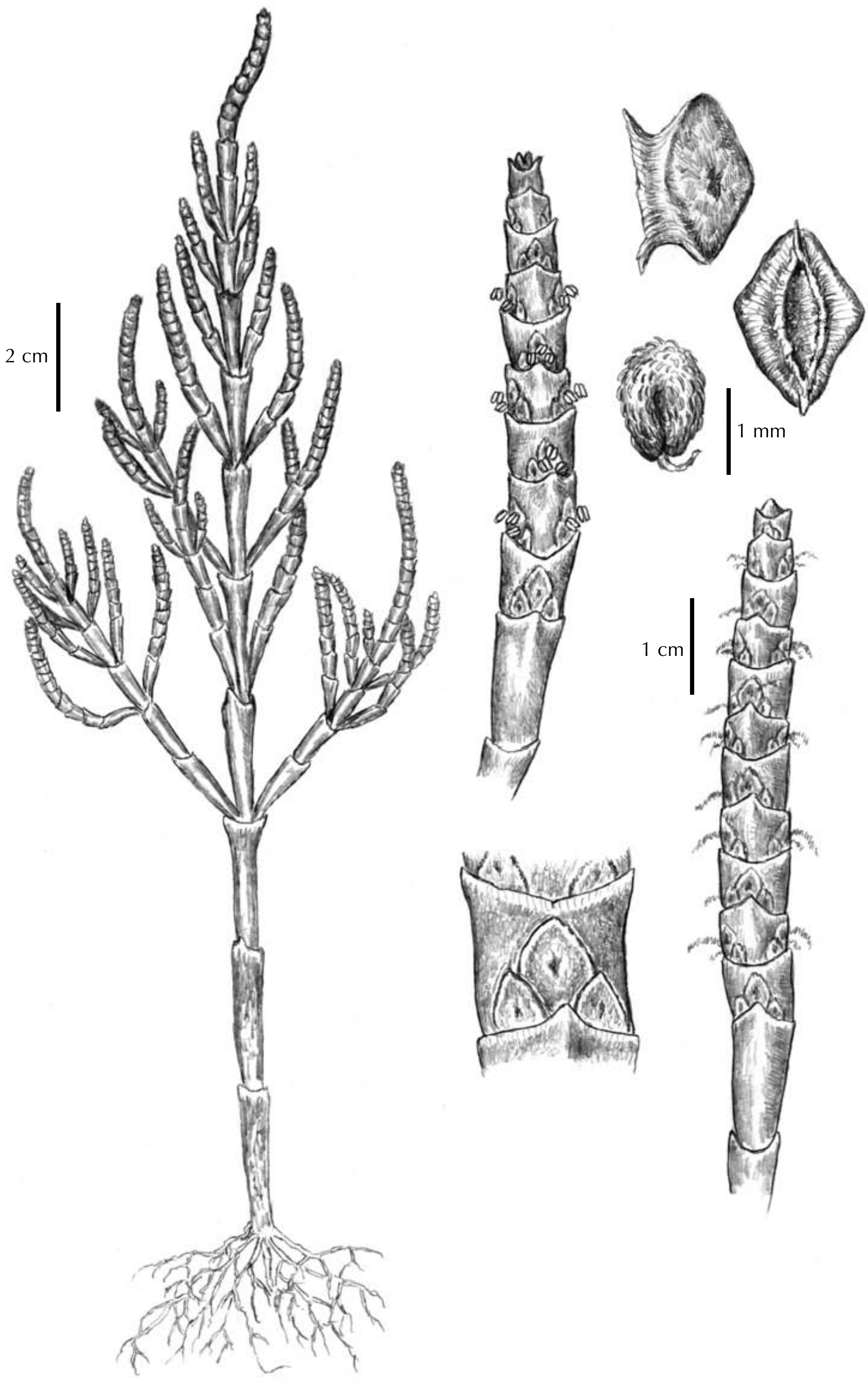

Plate 25. *Salicornia bigelovii*. McIntire Drawings, © 1999 by Zedler.

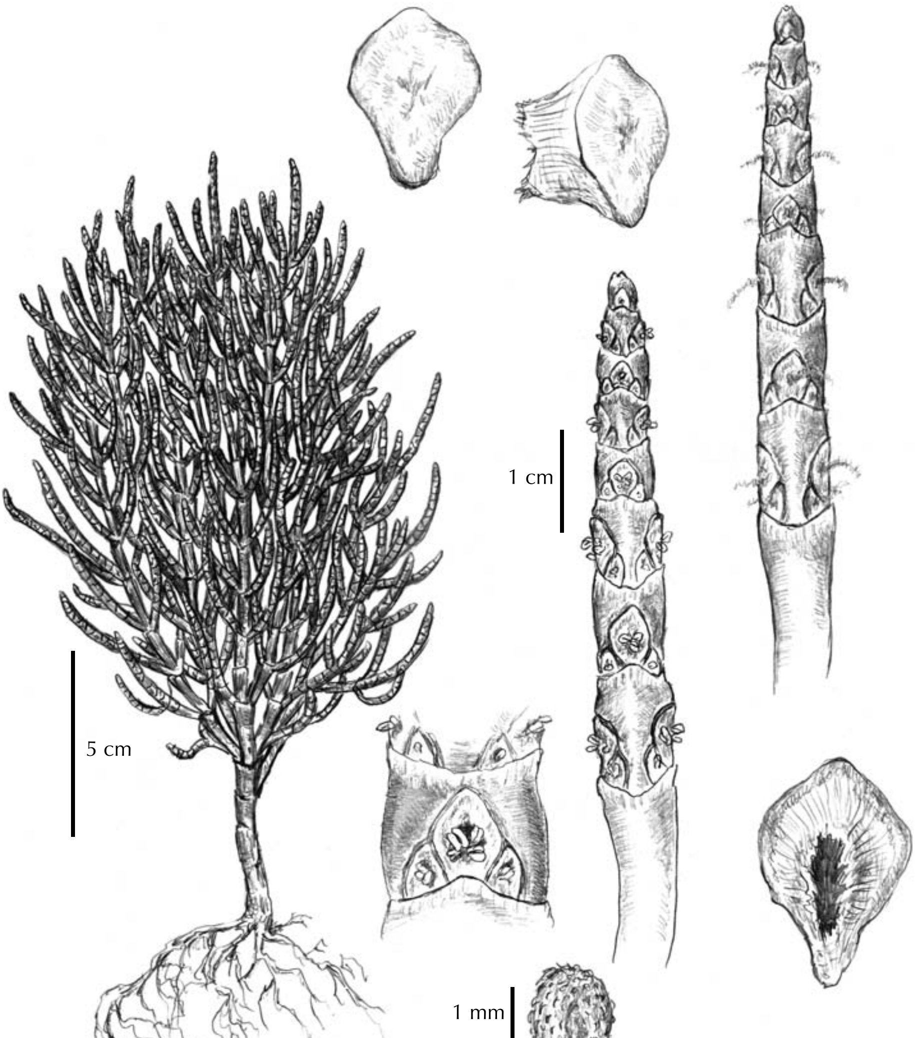

Plate 26. *Salicornia europaea*. McIntire Drawings, © 1999 by Zedler.

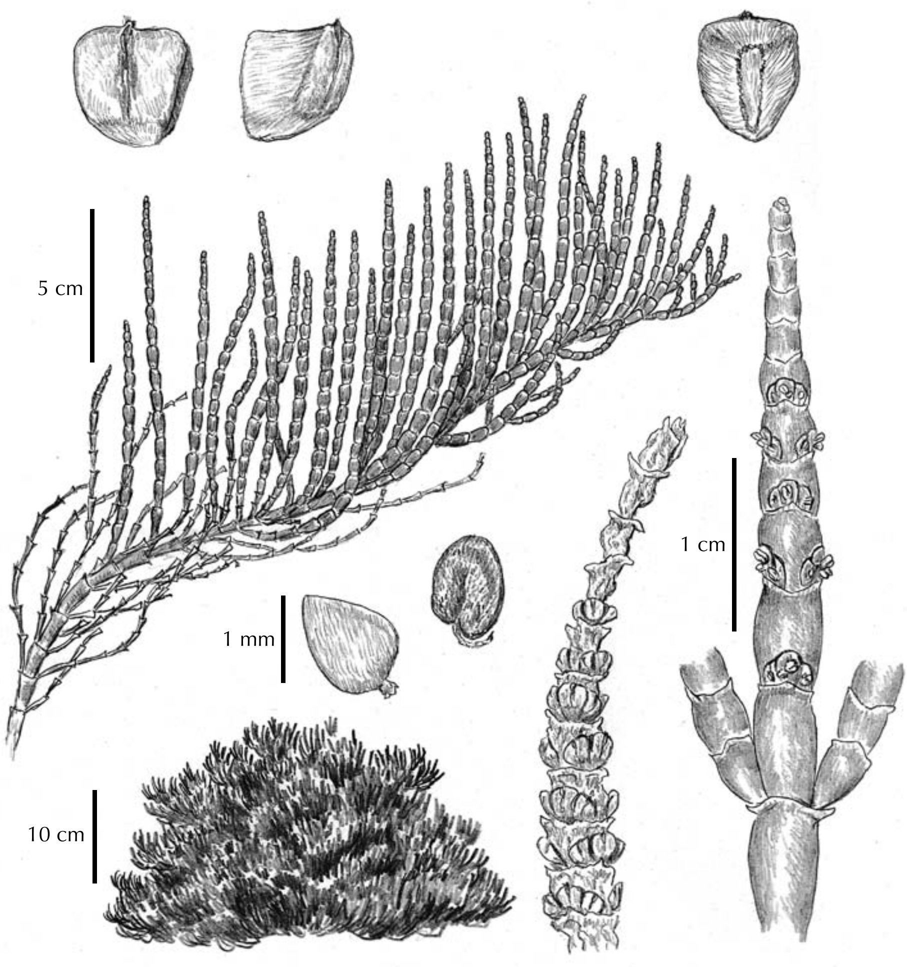

Plate 27. *Salicornia subterminalis*. McIntire Drawings, © 1999 by Zedler.

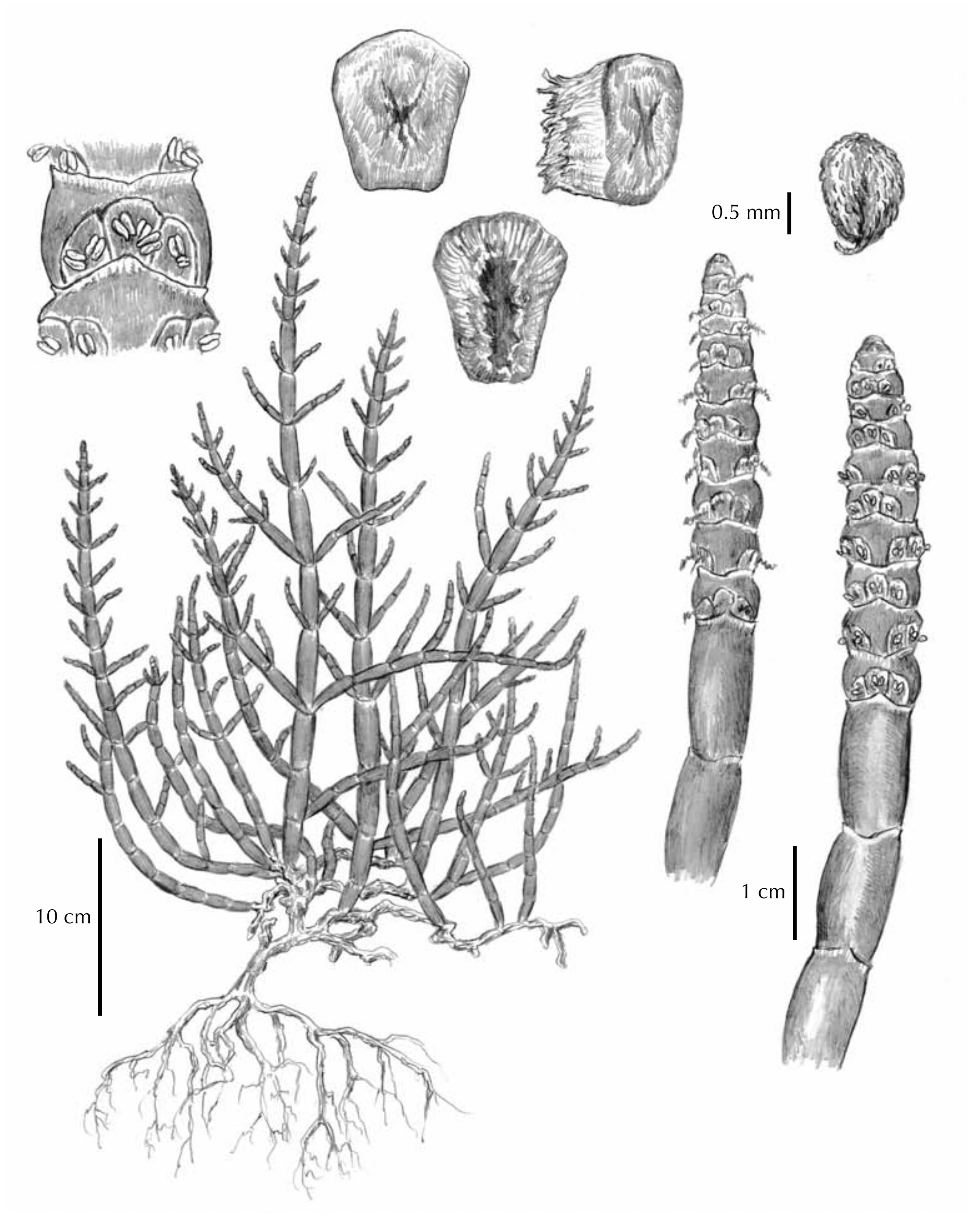

Plate 28. *Salicornia virginica*. McIntire Drawings, © 1999 by Zedler.

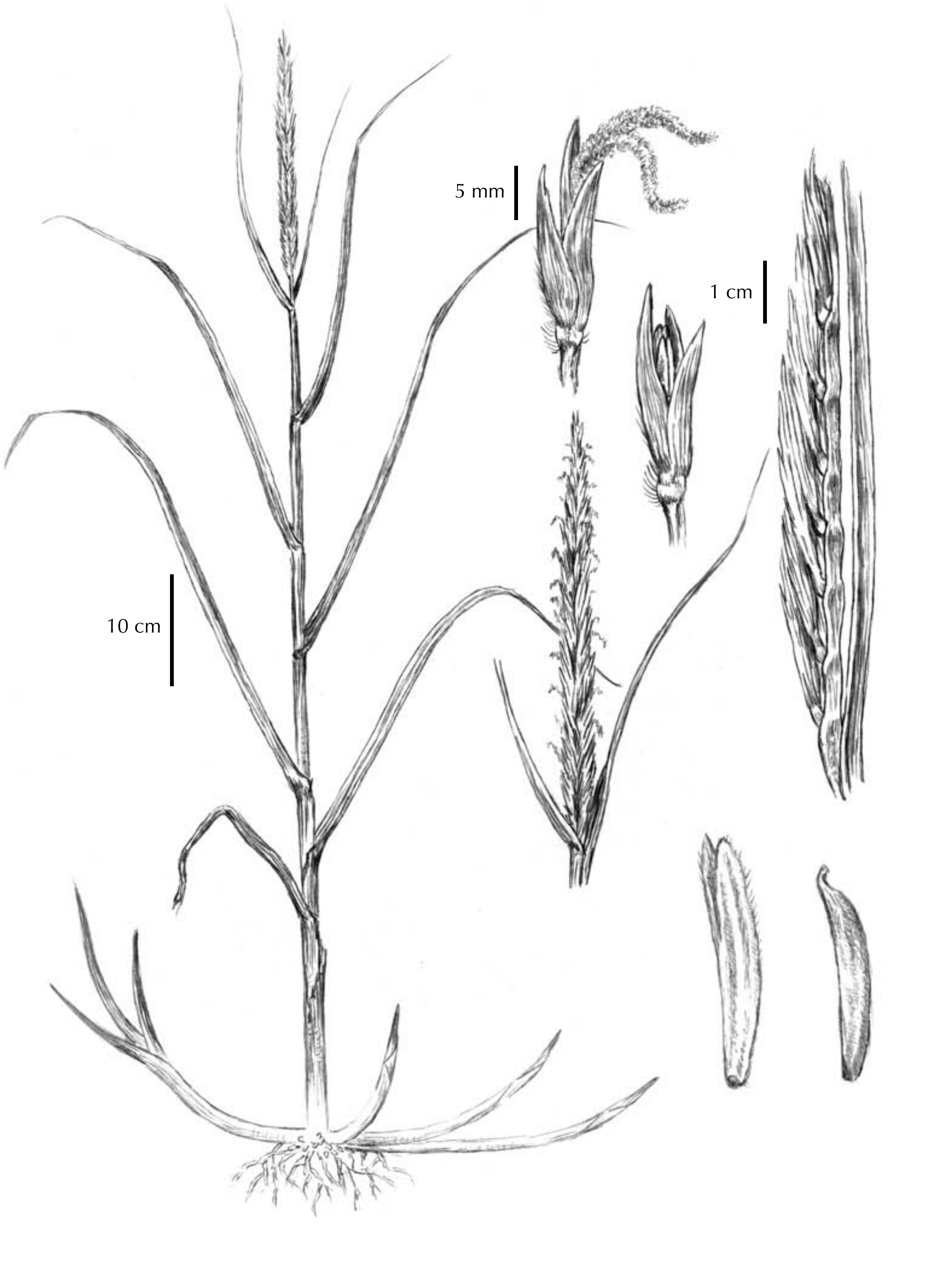

Plate 29. *Spartina foliosa*. McIntire Drawings, © 1999 by Zedler.

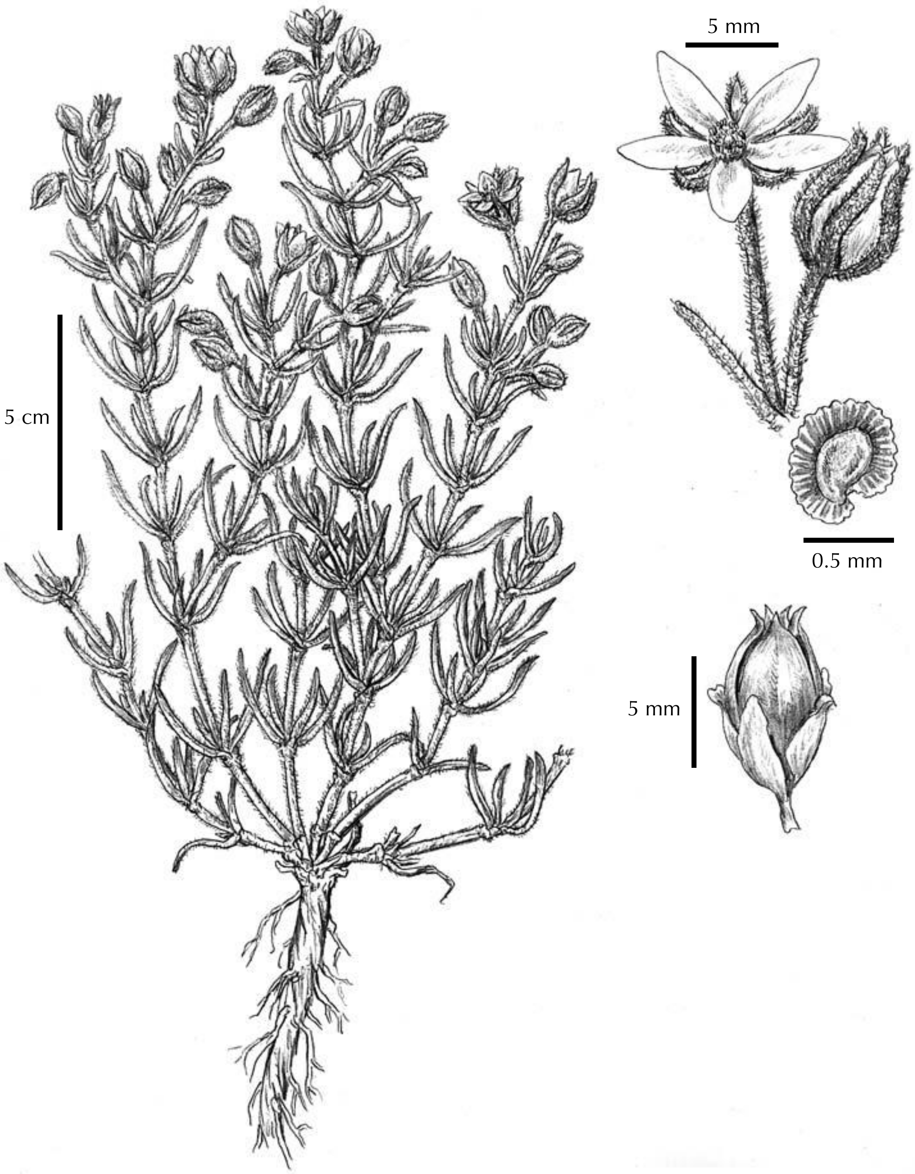

Plate 30. *Spergularia macrotheca*. McIntire Drawings, © 1999 by Zedler.

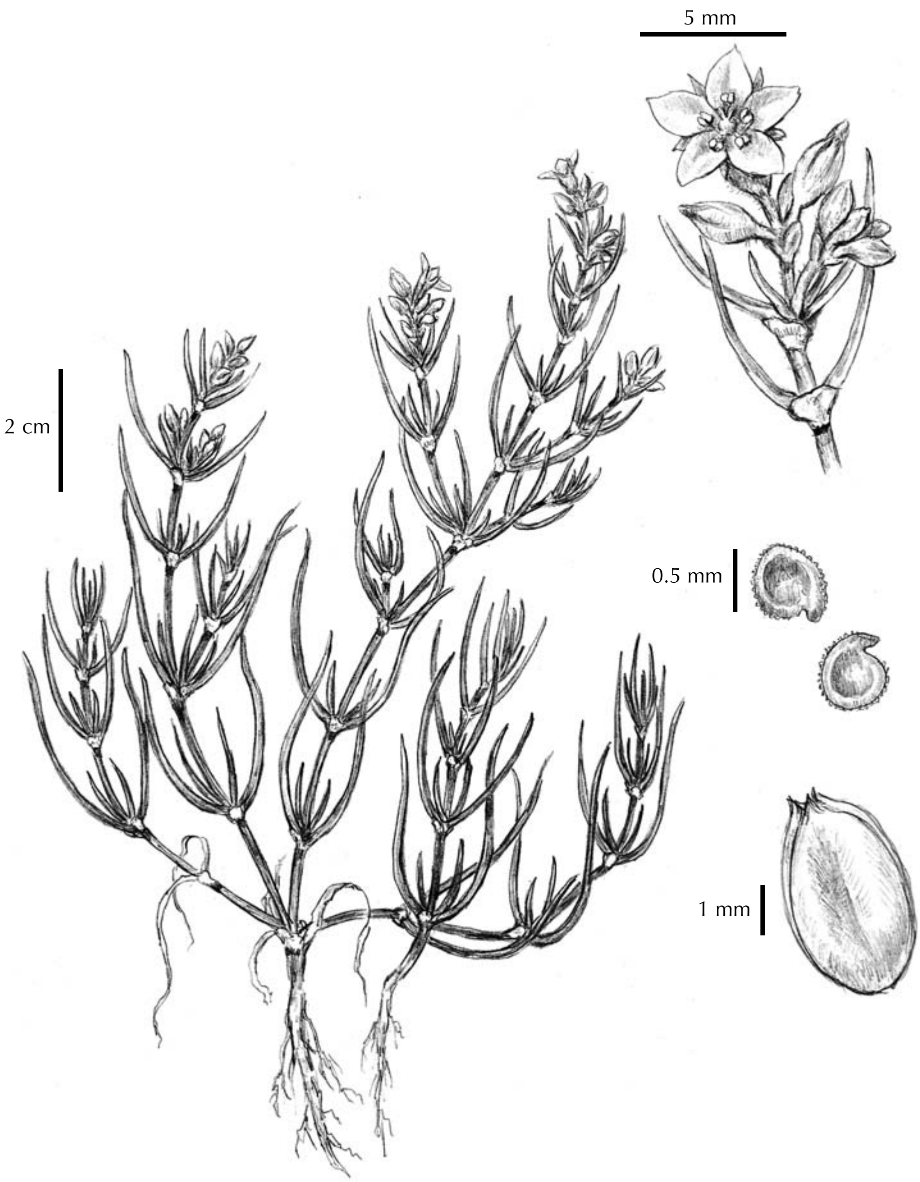

Plate 31. *Spergularia marina*. McIntire Drawings, © 1999 by Zedler.

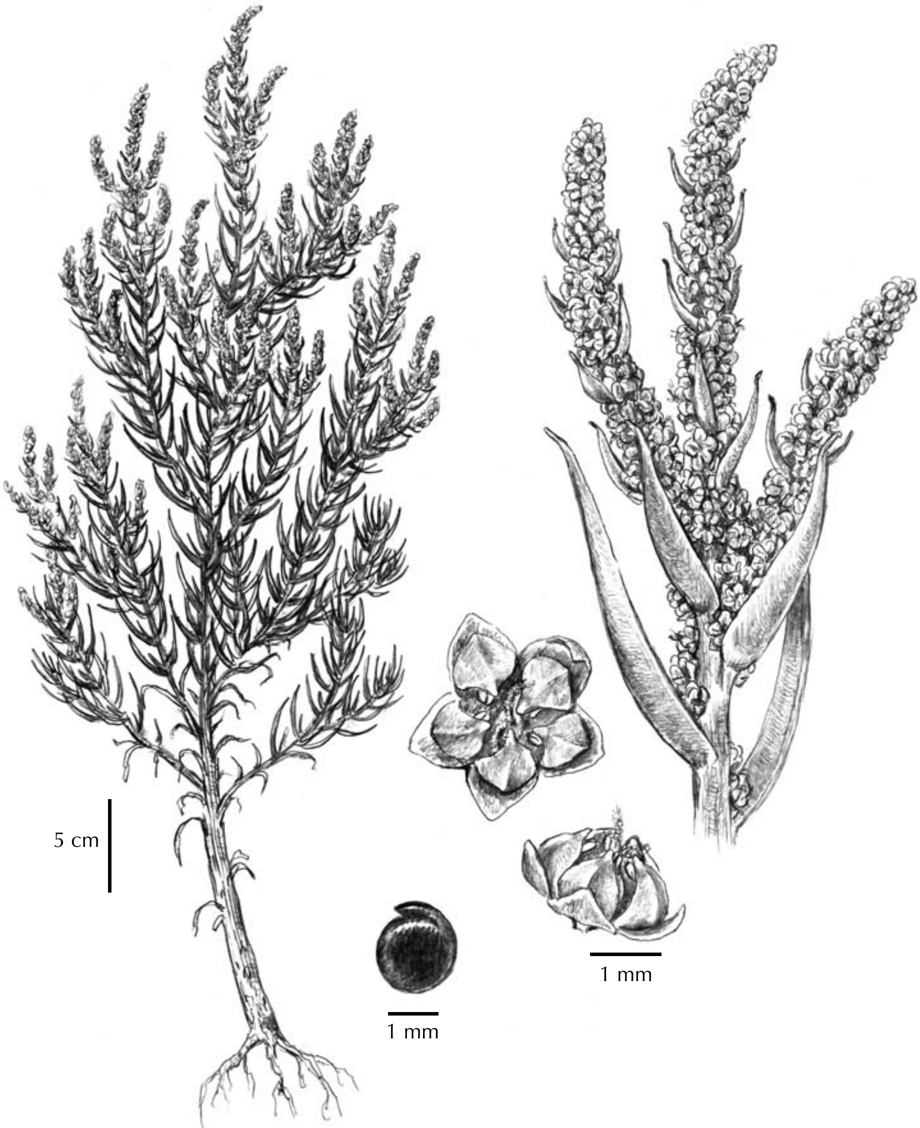

Plate 32. *Suaeda calceoliformis*. McIntire Drawings, © 1999 by Zedler.

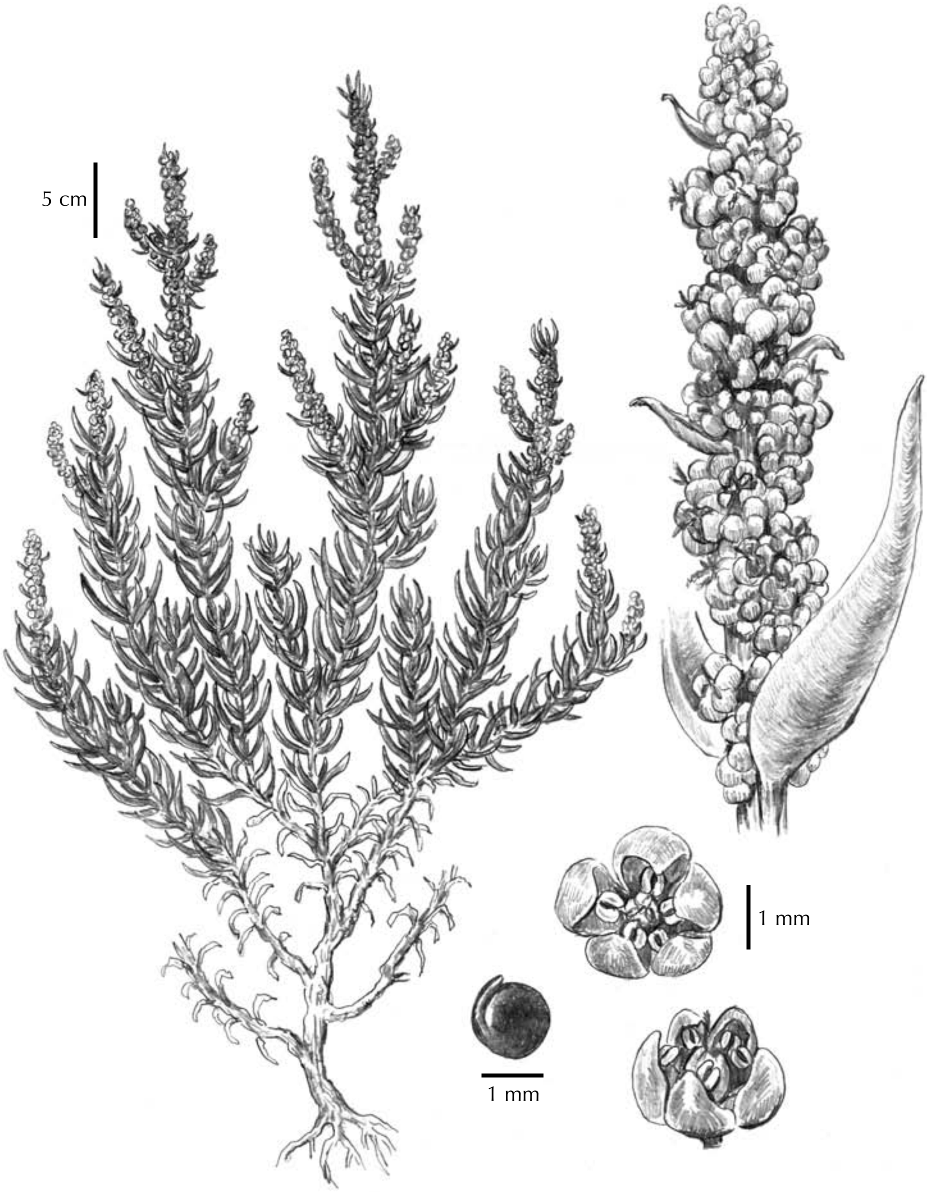

Plate 33. *Suaeda esteroa*. McIntire Drawings, © 1999 by Zedler.

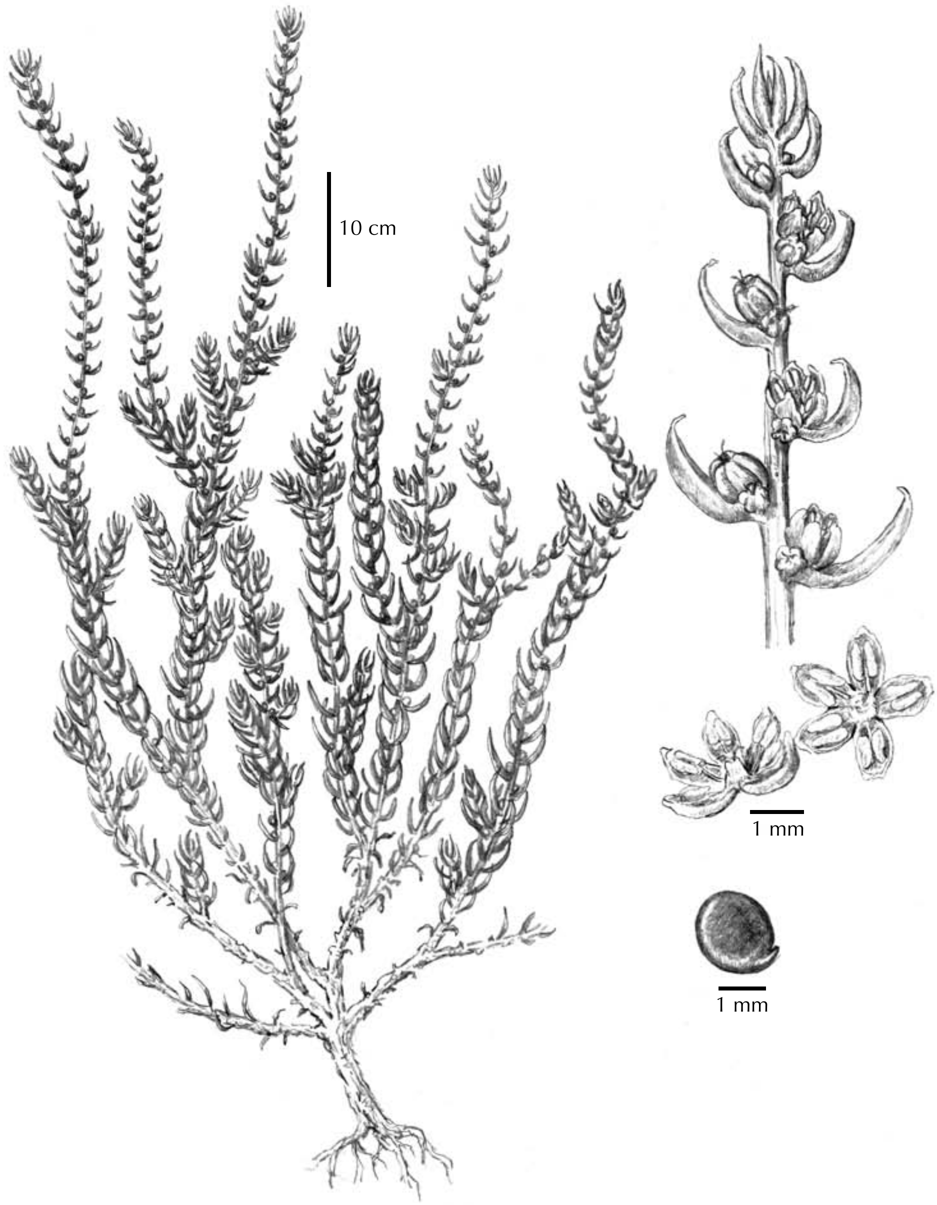

Plate 34. *Suaeda moquinii*. McIntire Drawings, © 1999 by Zedler.

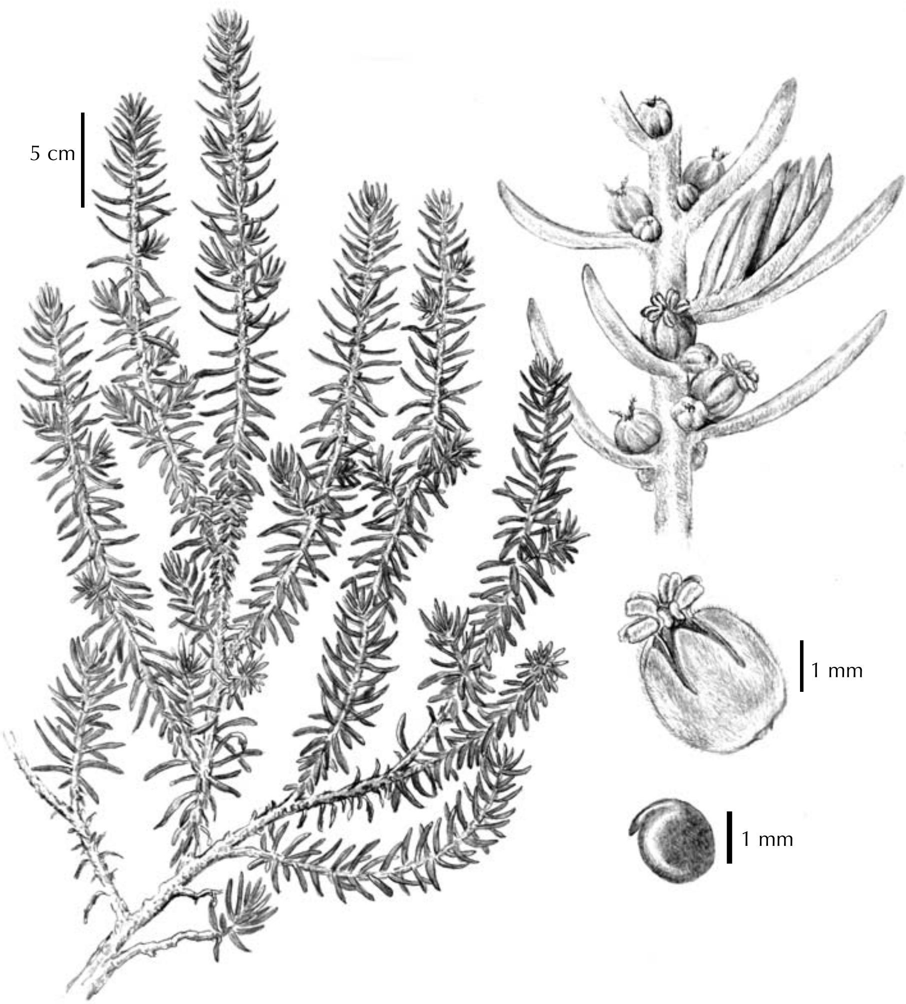

Plate 35. *Suaeda taxifolia*. McIntire Drawings, © 1999 by Zedler.

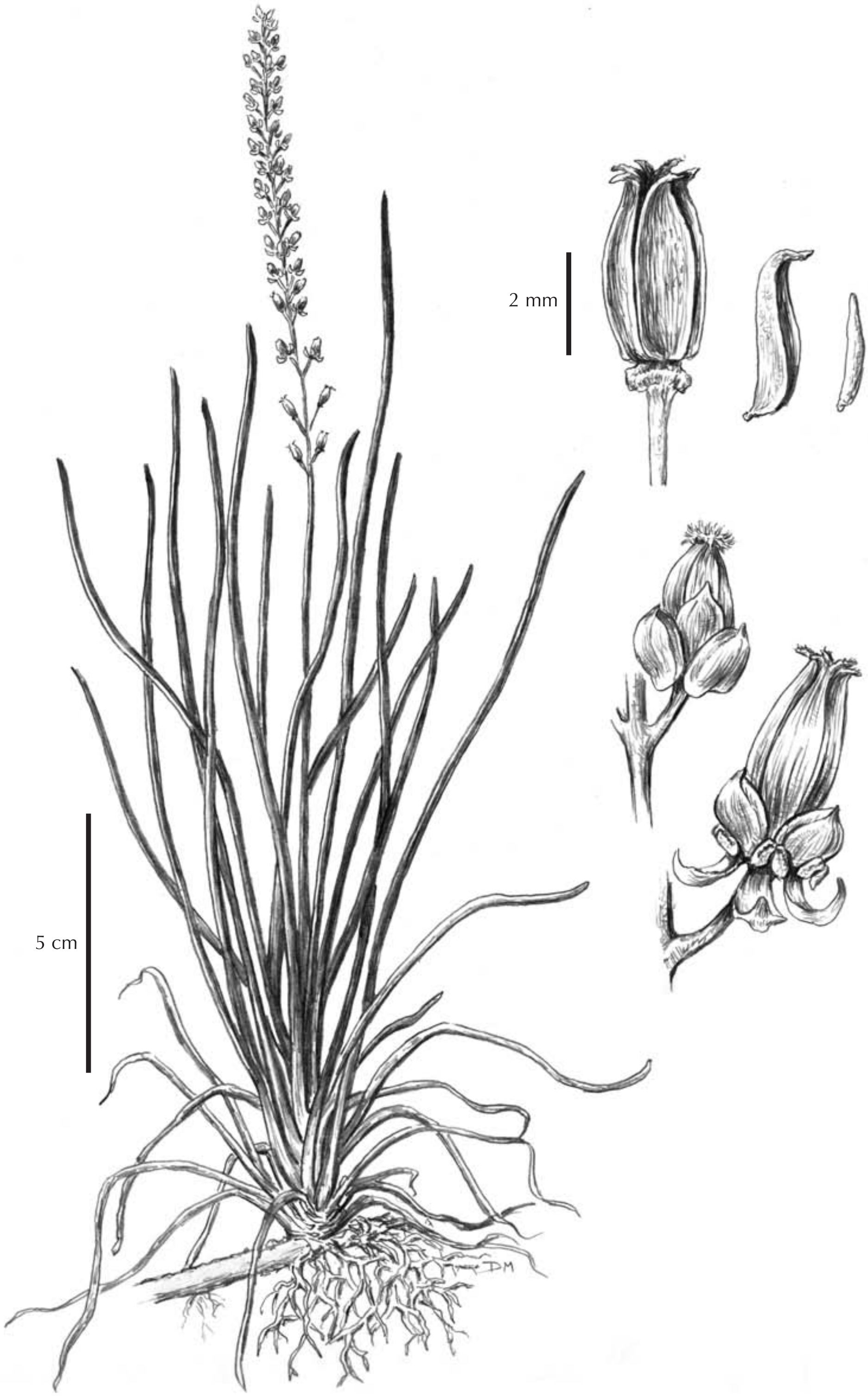

Plate 36. *Triglochin concinna*. McIntire Drawings, © 1999 by Zedler.

Plate 37. *Zostera marina*. McIntire Drawings, © 1999 by Zedler.

Plate 38. *Atriplex semibaccata*. McIntire Drawings, © 1999 by Zedler.

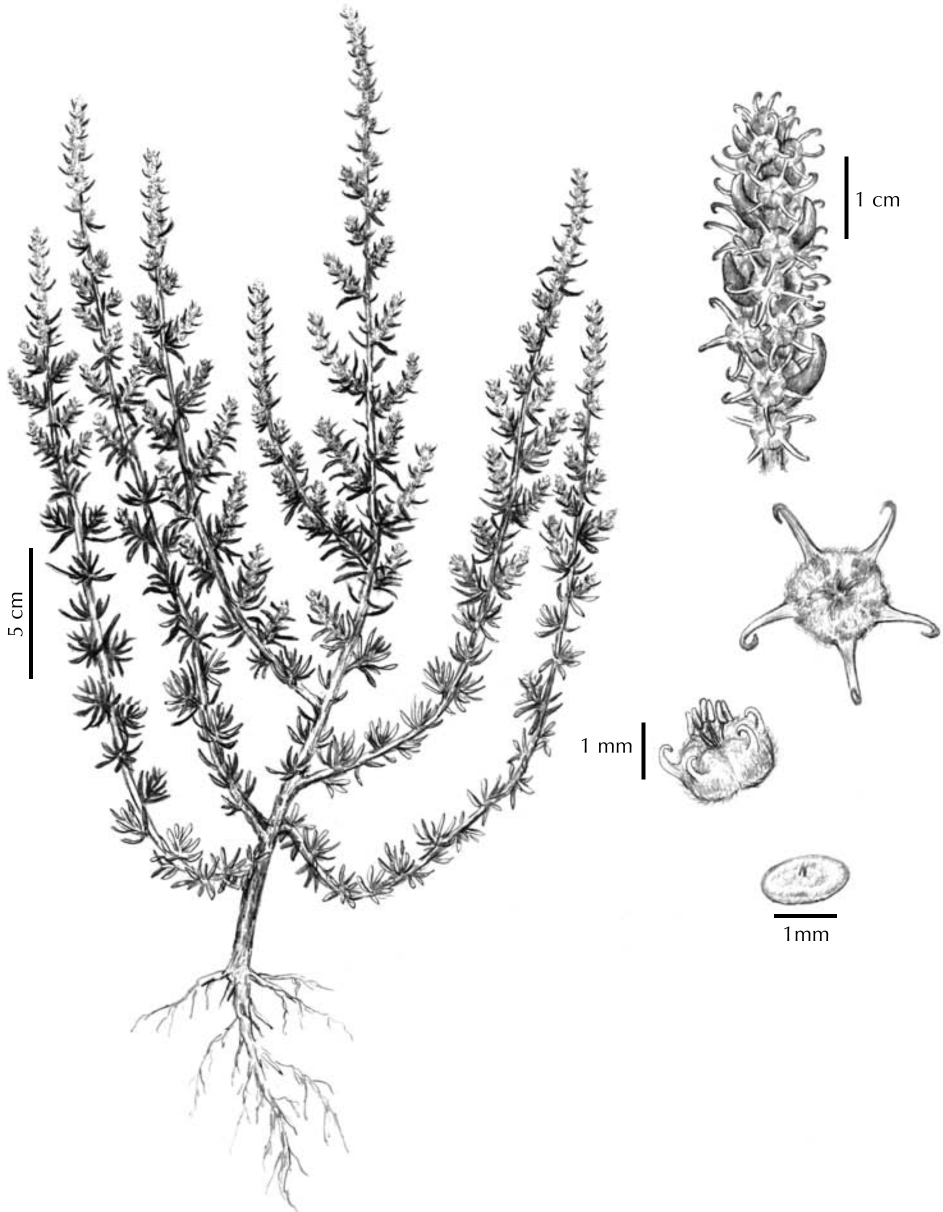

Plate 39. *Bassia hyssopifolia*. McIntire Drawings, © 1999 by Zedler.

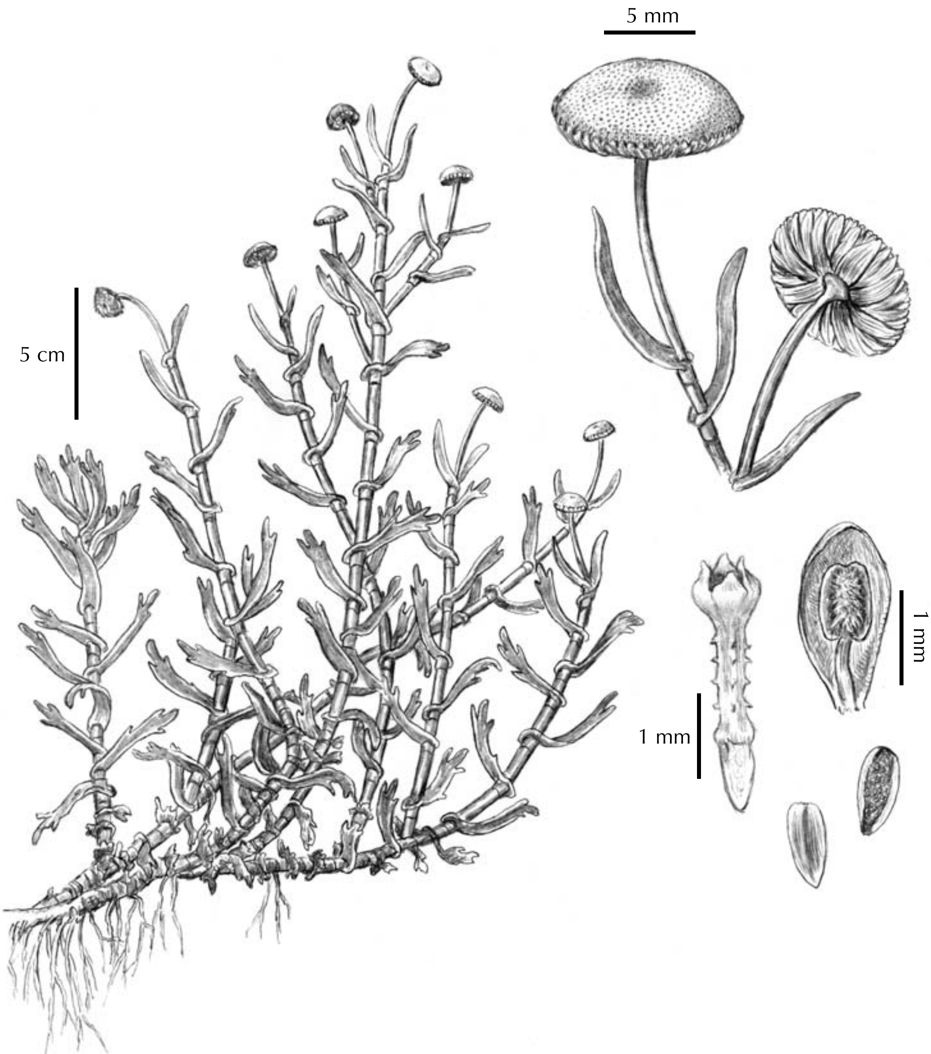

Plate 40. *Cotula coronopifolia*. McIntire Drawings, © 1999 by Zedler.

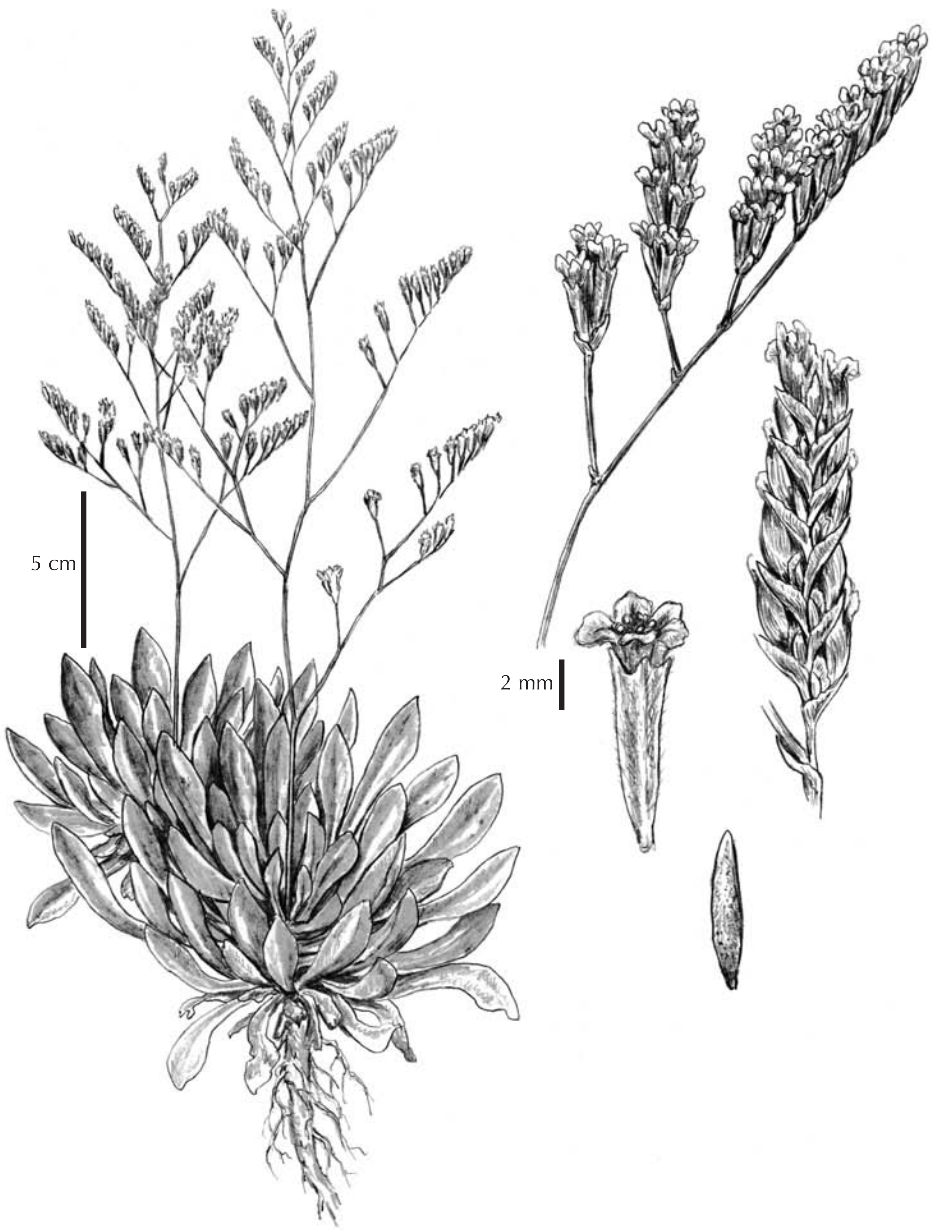

Plate 41. *Limonium ramosissimum* ssp. *provinciale*. McIntire Drawings, © 1999 by Zedler.

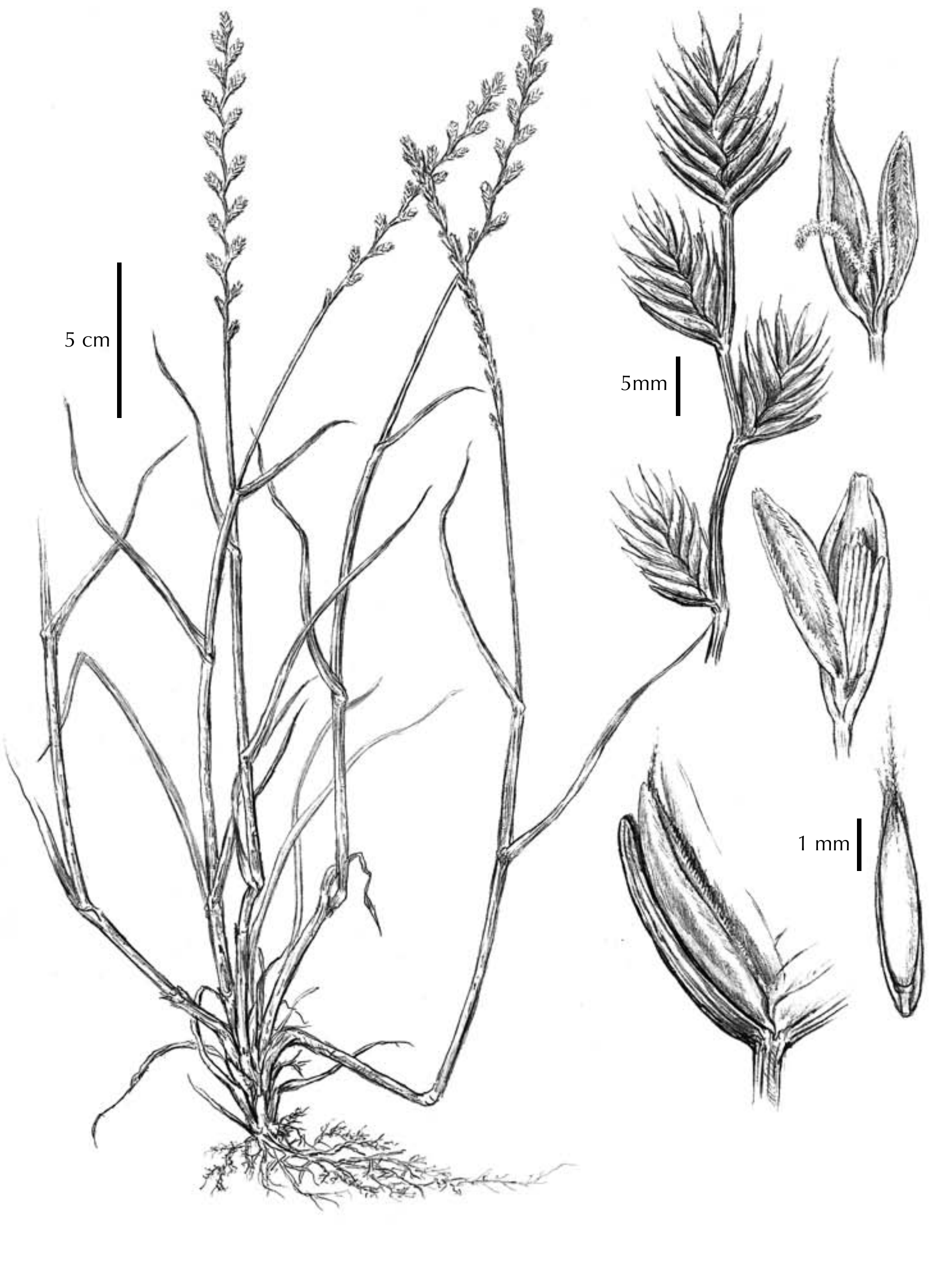

Plate 42. *Lolium multiflorum*. McIntire Drawings, © 1999 by Zedler.

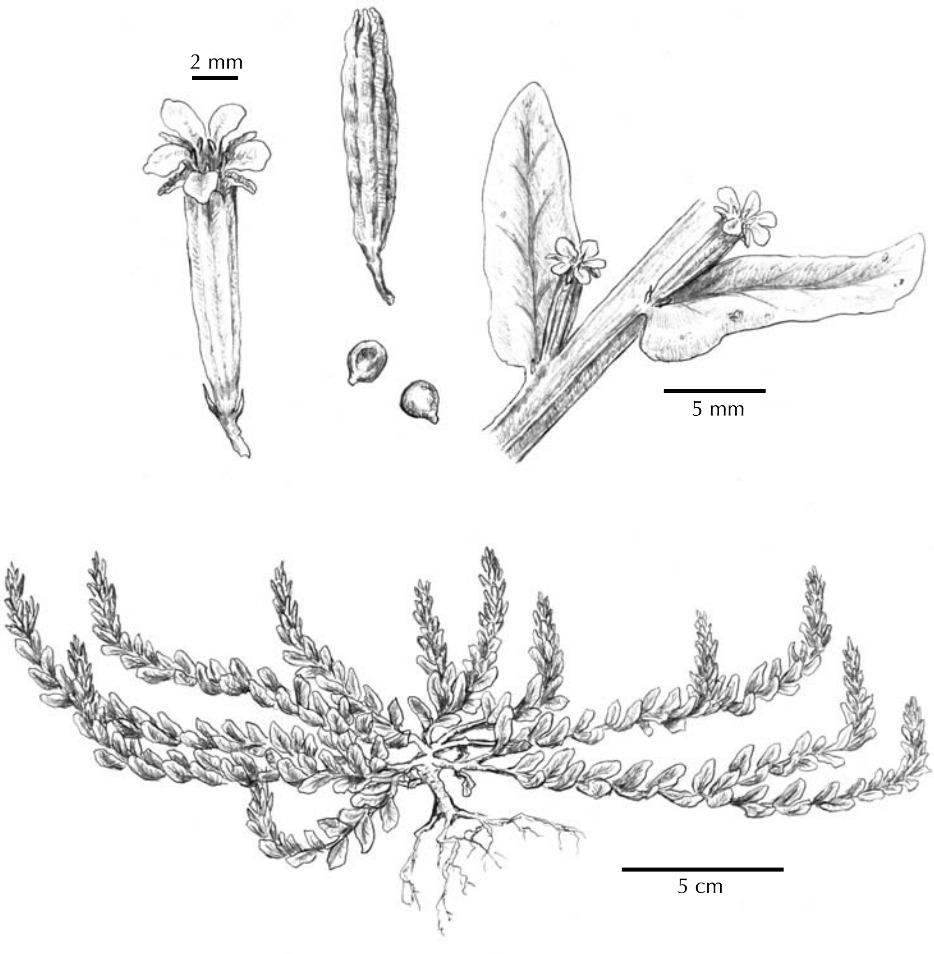

Plate 43. *Lythrum hyssopifolium*. McIntire Drawings, © 1999 by Zedler.

Plate 44. *Mesembryanthemum crystallinum*. McIntire Drawings, © 1999 by Zedler.

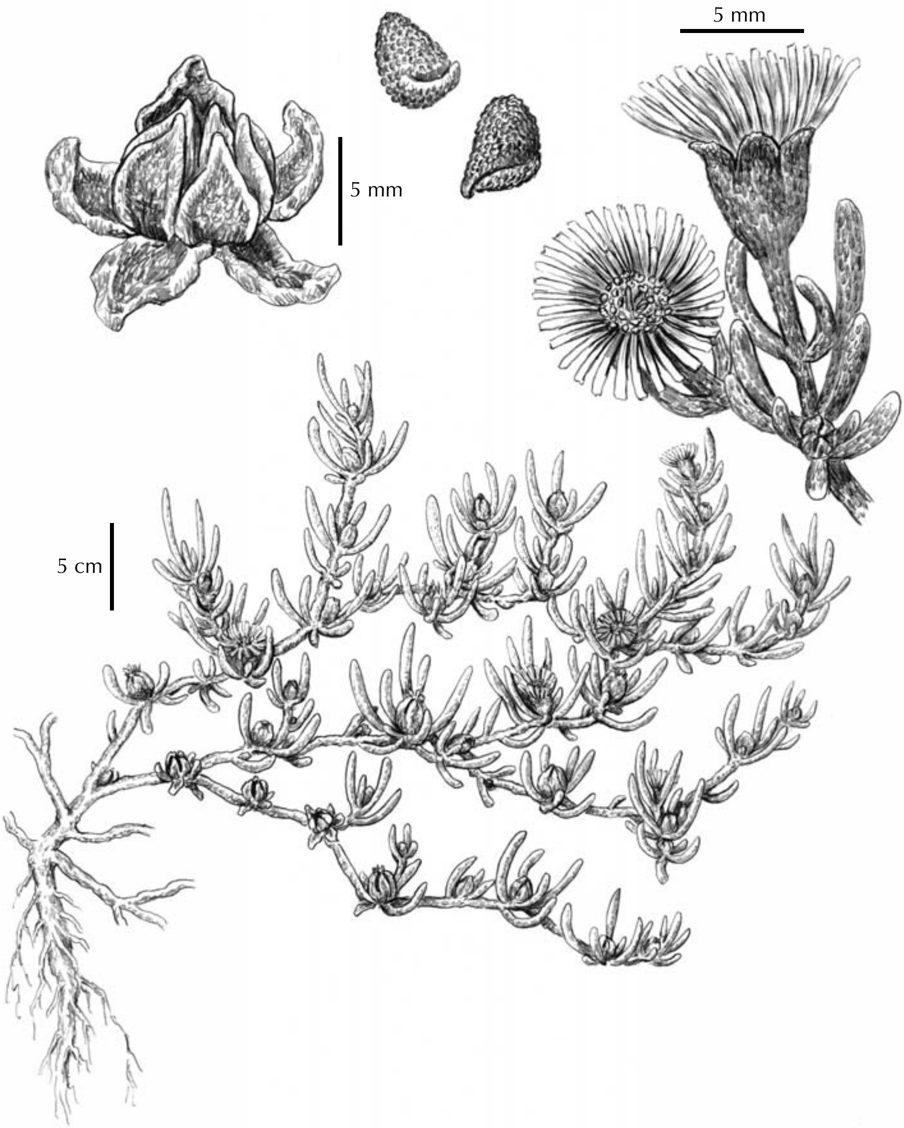

Plate 45. *Mesembryanthemum nodiflorum*. McIntire Drawings, © 1999 by Zedler.

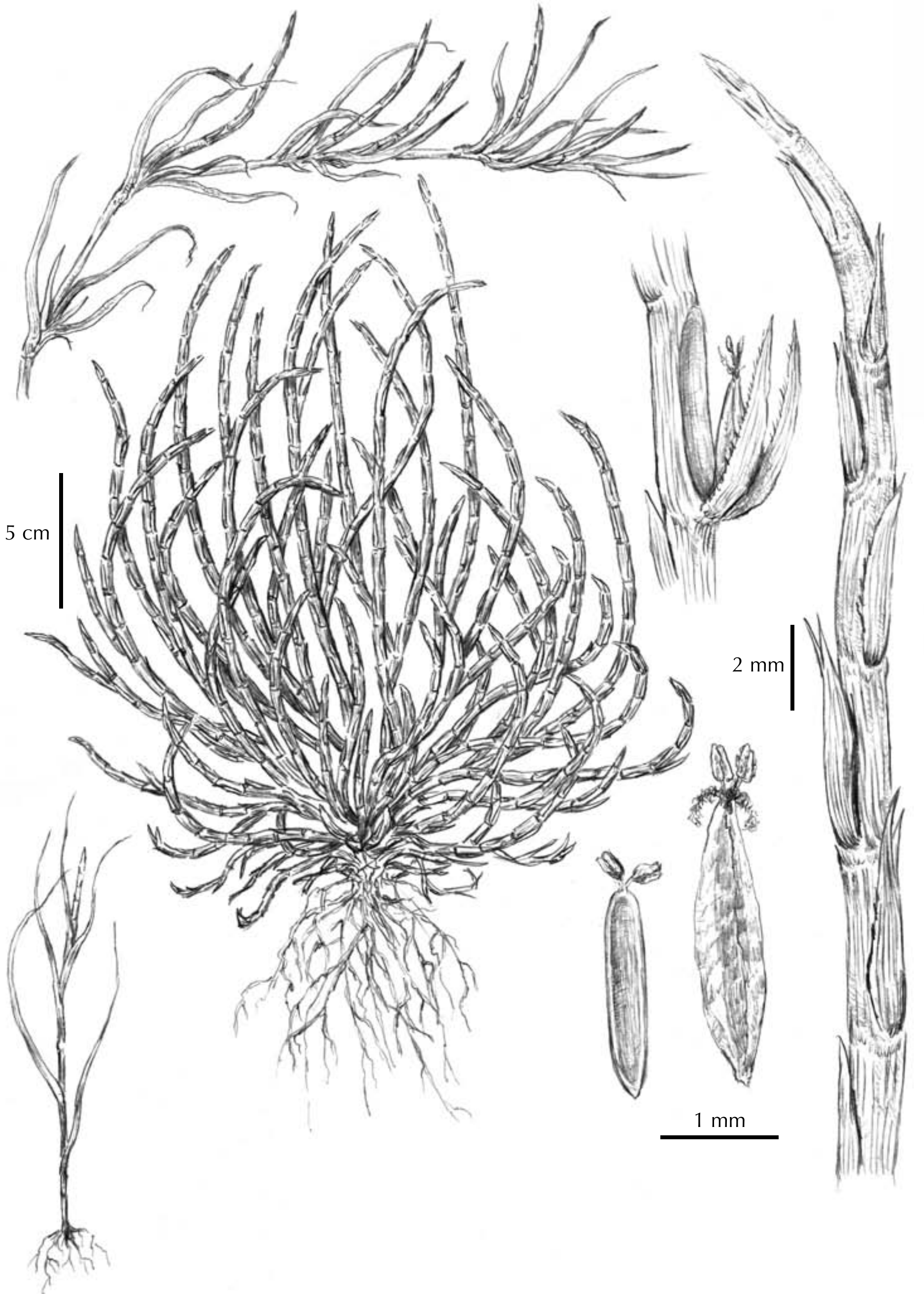

Plate 46. *Parapholis incurva*. McIntire Drawings, © 1999 by Zedler.

Plate 47. *Polypogon monspeliensis*. McIntire Drawings, © 1999 by Zedler.

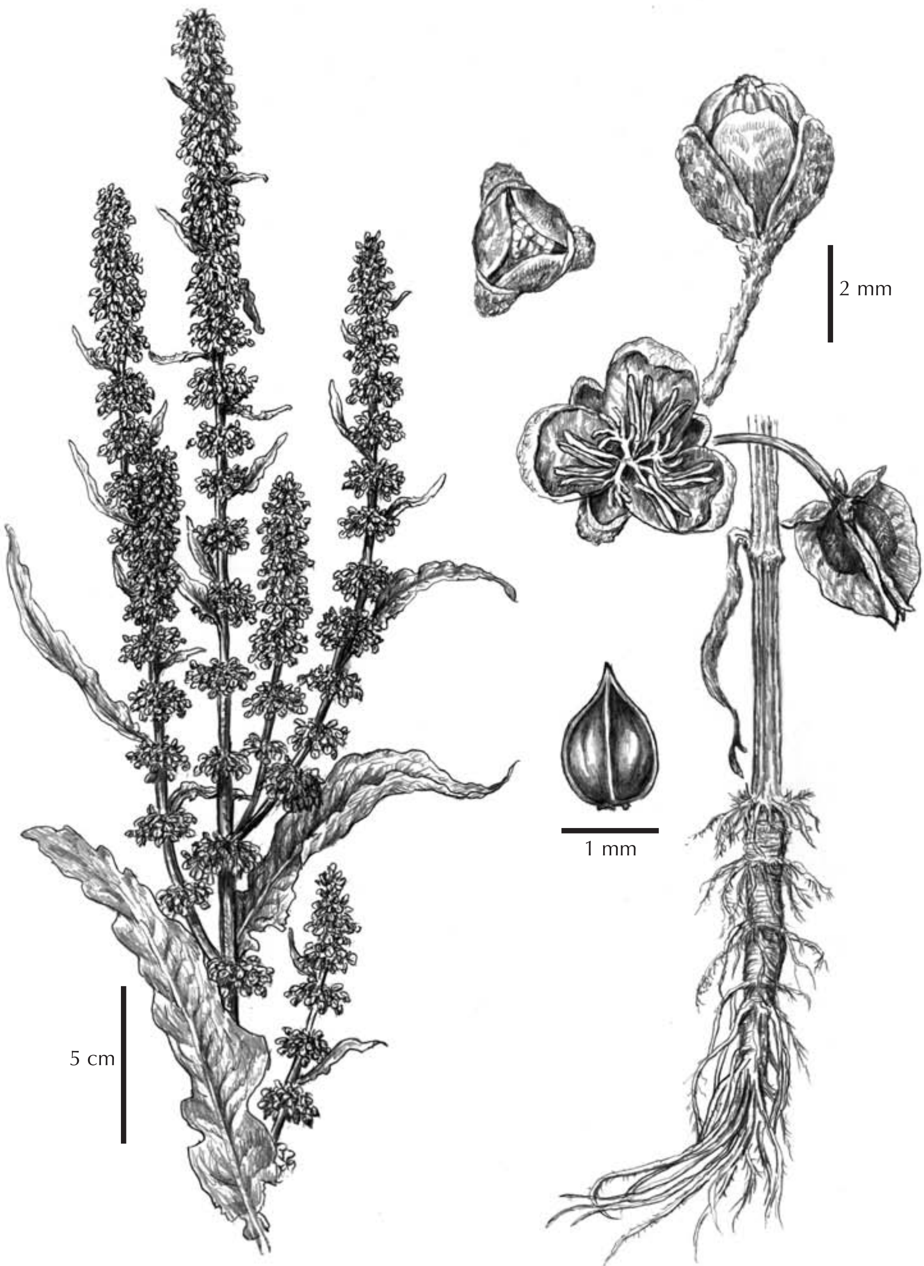

Plate 48. *Rumex crispus*. McIntire Drawings, © 1999 by Zedler.

Plate 49. *Sonchus asper*. McIntire Drawings, © 1999 by Zedler.

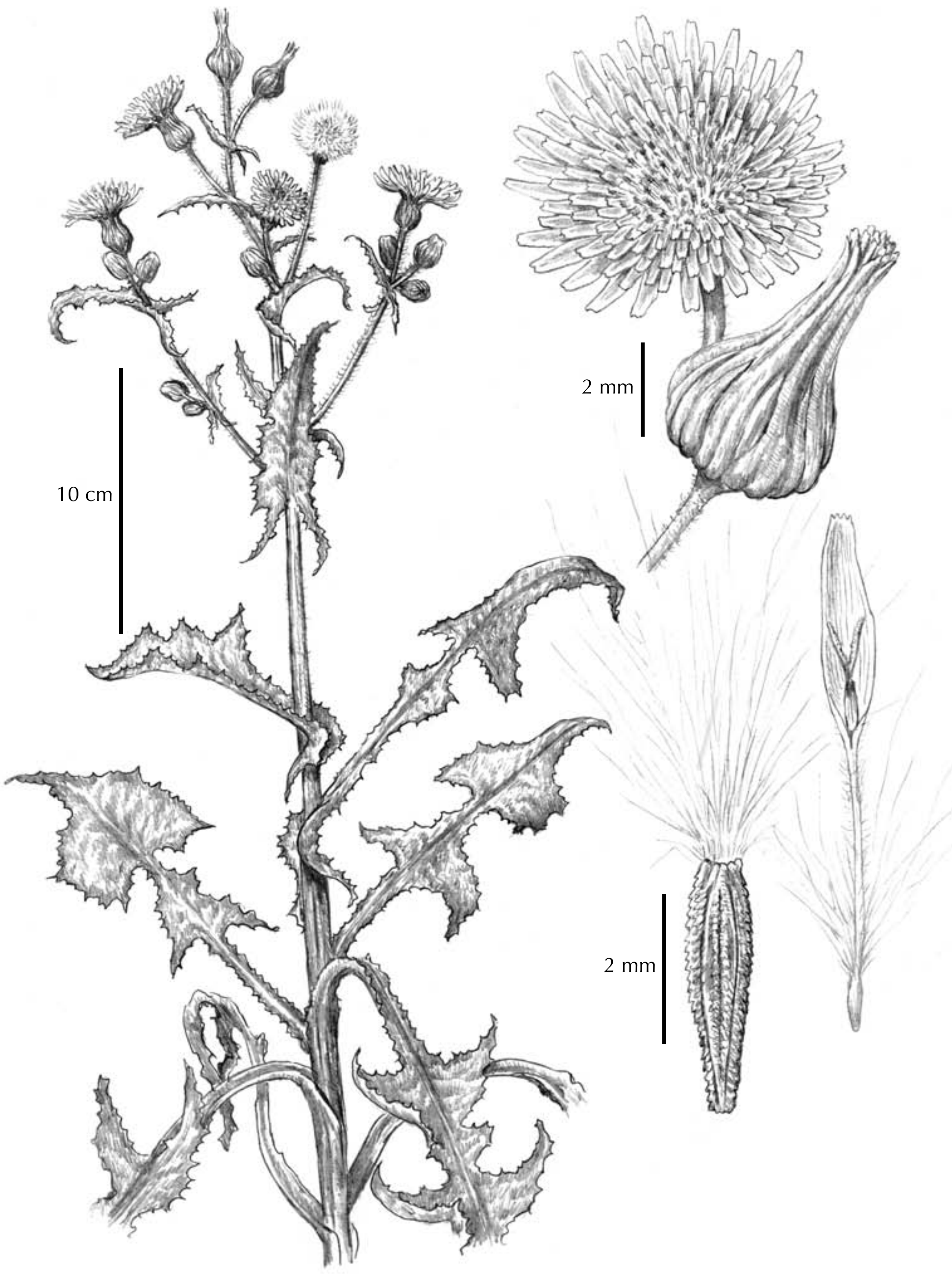

Plate 50. *Sonchus oleraceus*. McIntire Drawings, © 1999 by Zedler.

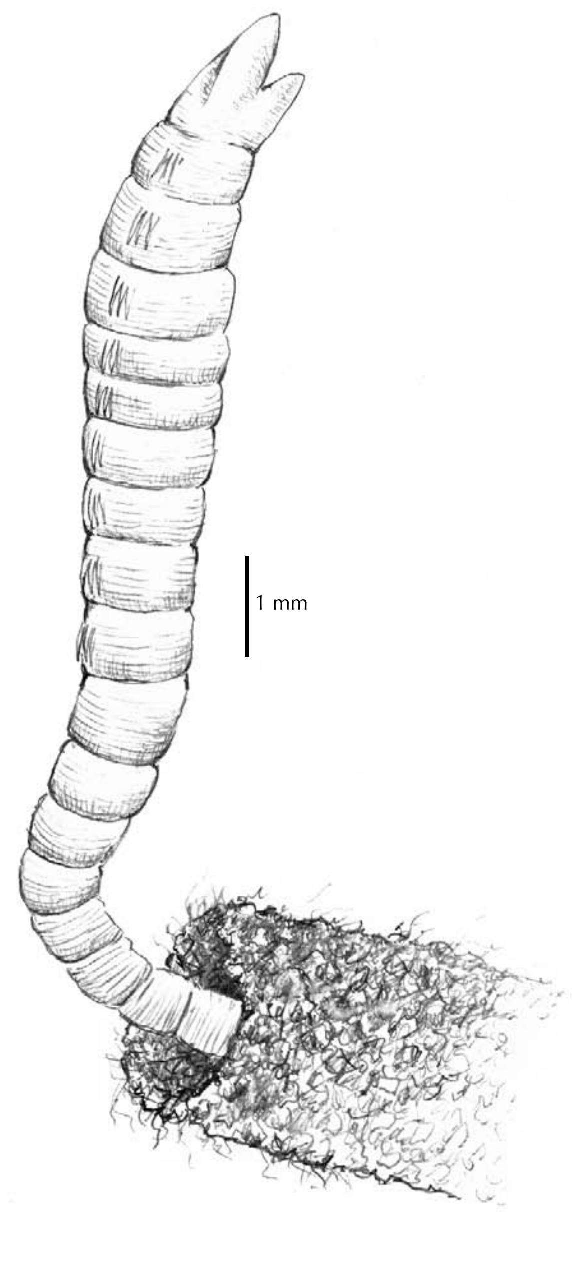

Plate 51. *Capitella capitata*. McIntire Drawings, © 1999 by Zedler.

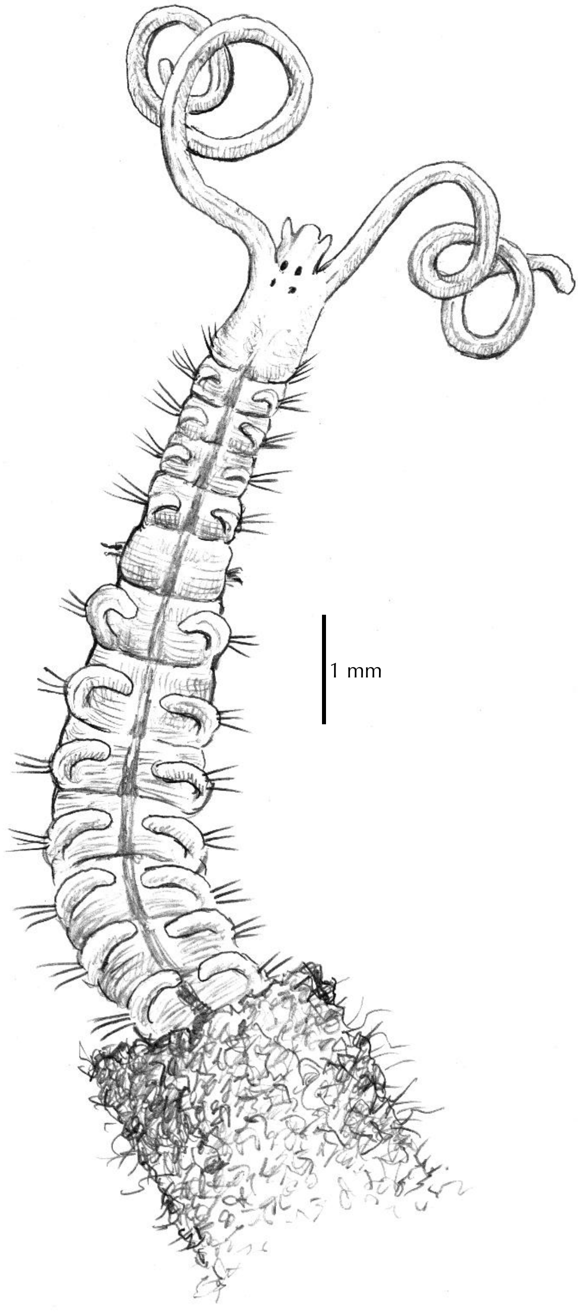

Plate 52. *Polydora nuchalis*. McIntire Drawings, © 1999 by Zedler.

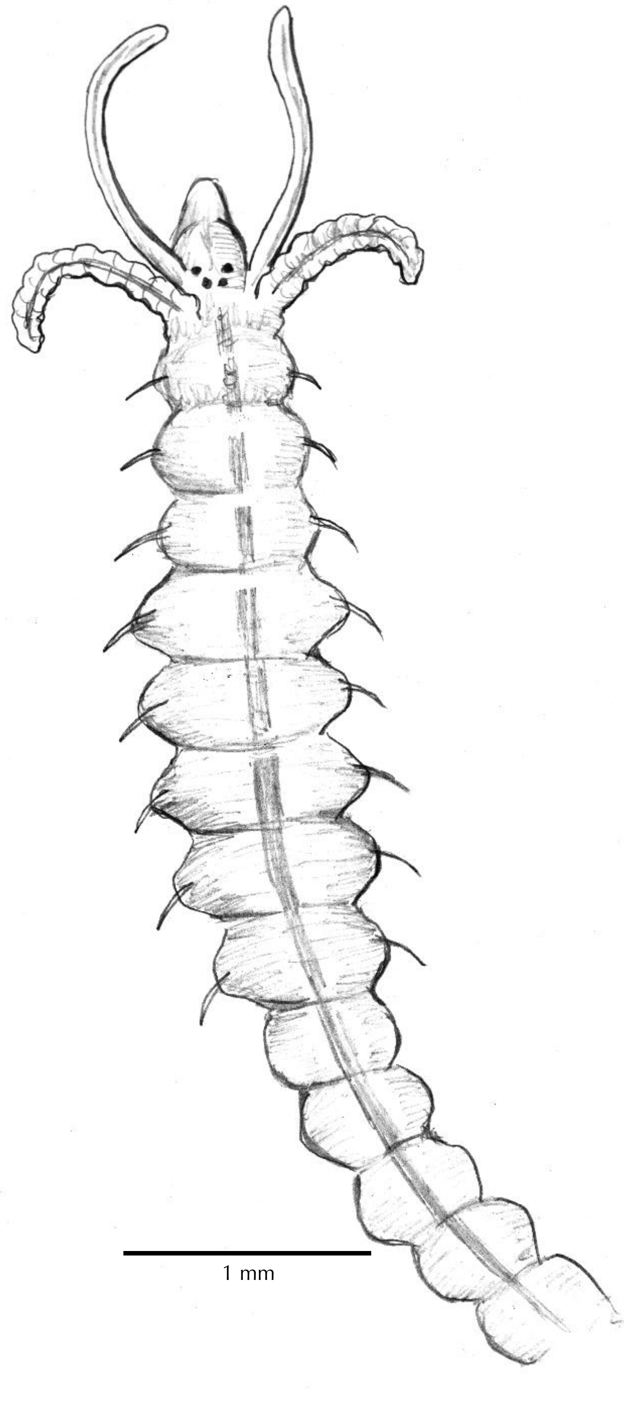

Plate 53. *Streblospio benedicti*. McIntire Drawings, © 1999 by Zedler.

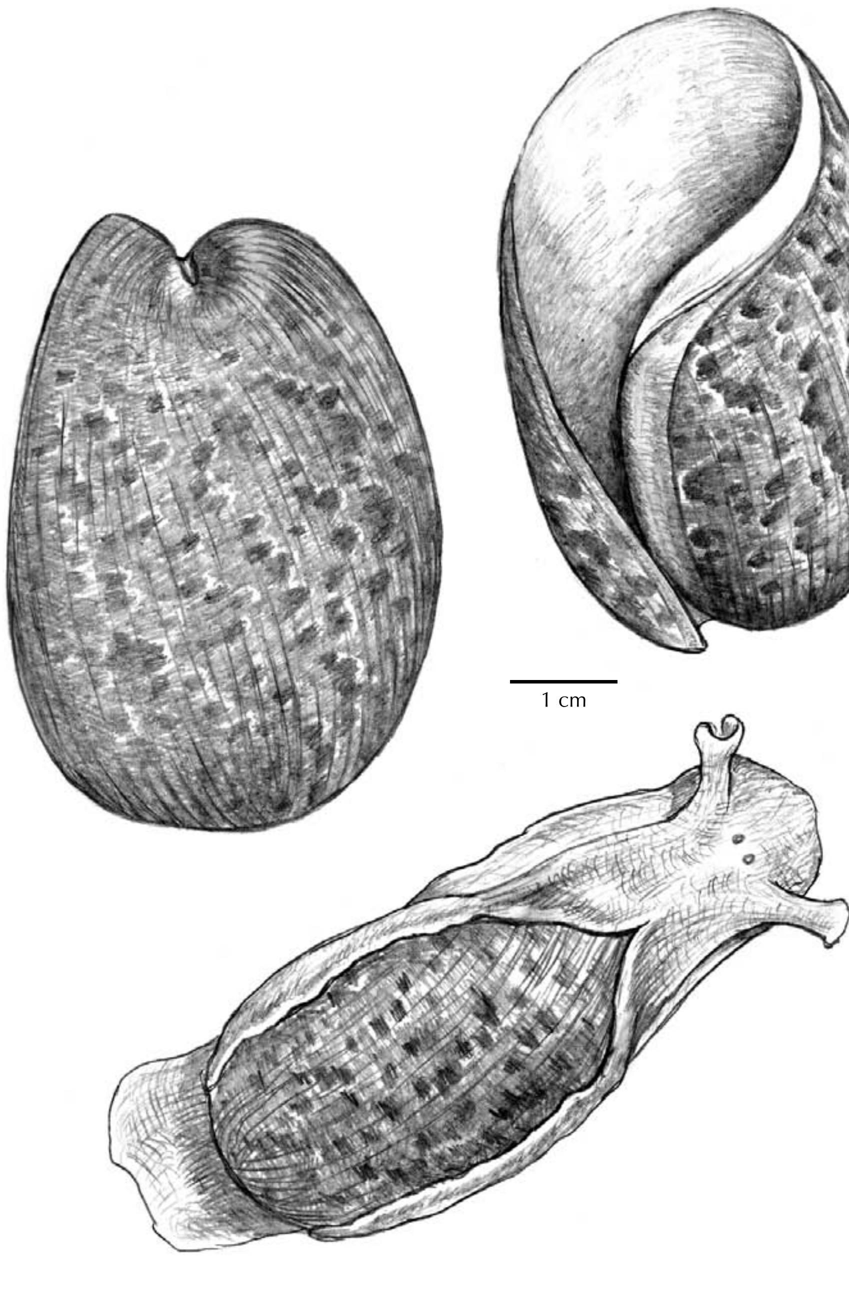

Plate 54. *Bulla gouldiana*. McIntire Drawings, © 1999 by Zedler.

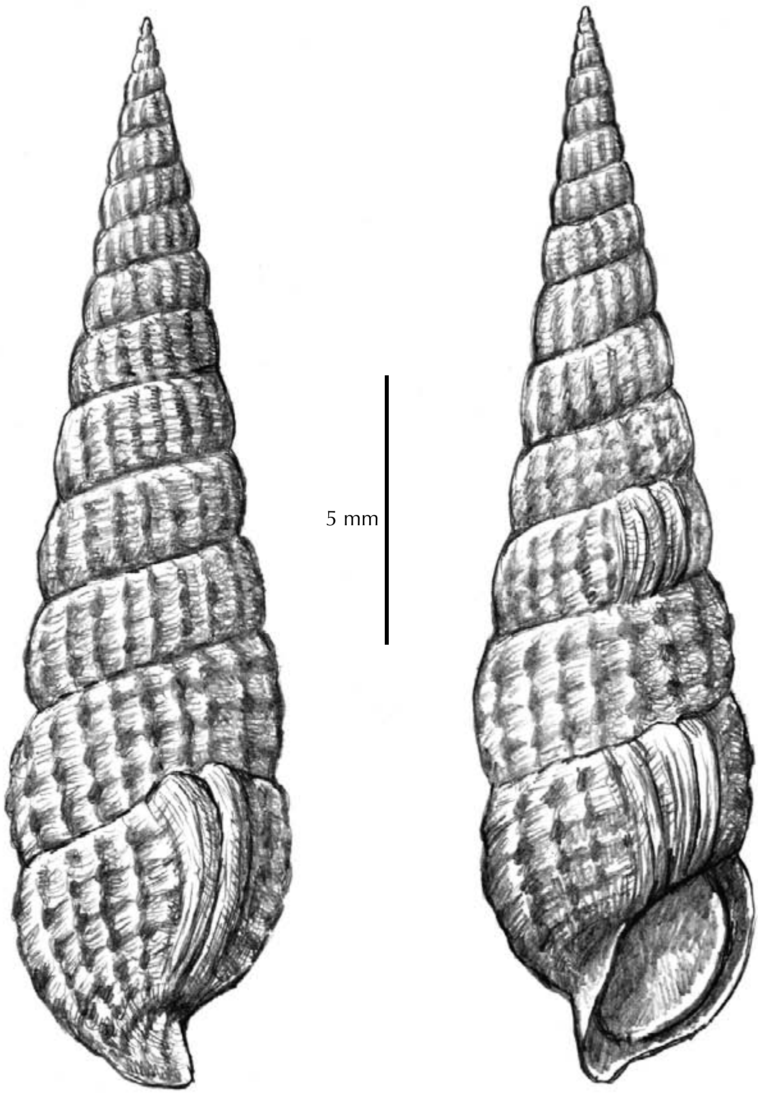

Plate 55. *Cerithidea californica*. McIntire Drawings, © 1999 by Zedler.

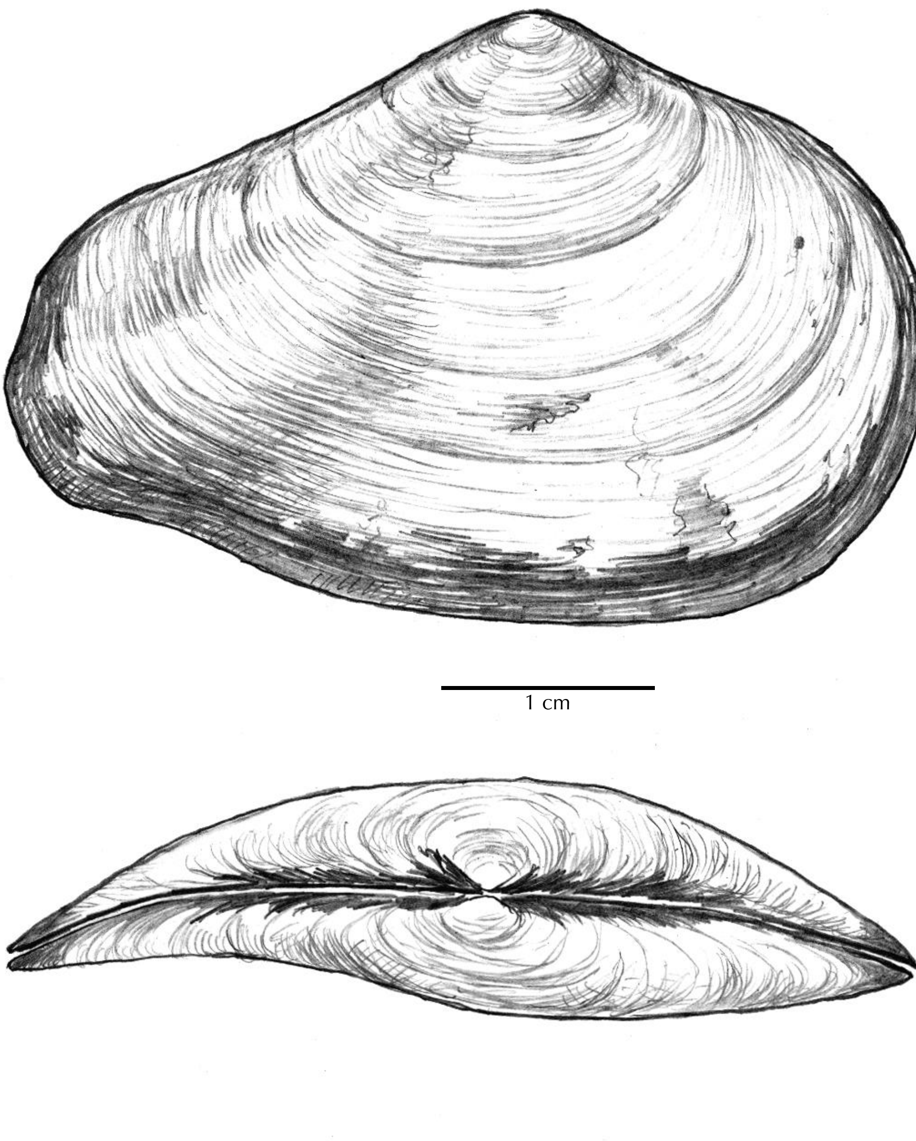

Plate 56. *Macoma nasuta*. McIntire Drawings, © 1999 by Zedler.

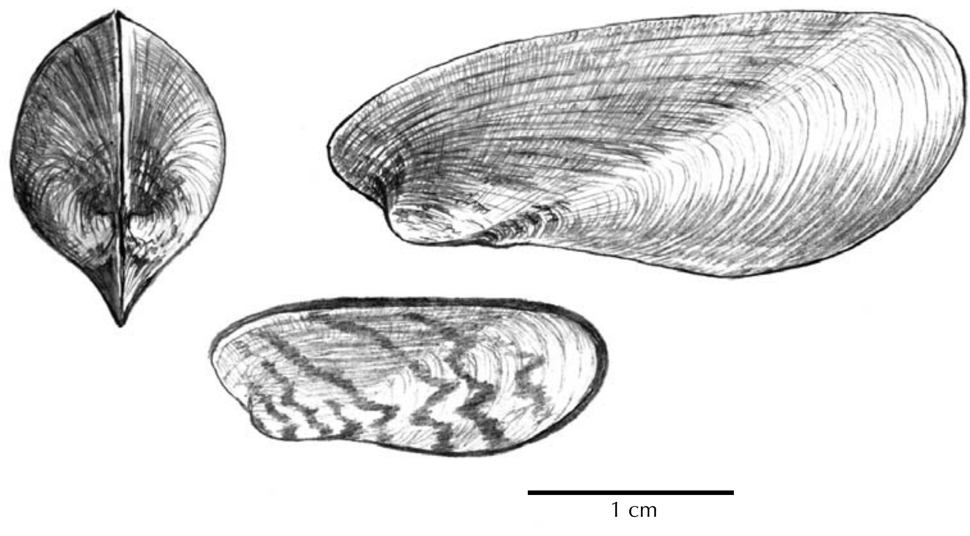

Plate 57. *Musculista senhousia*. McIntire Drawings, © 1999 by Zedler.

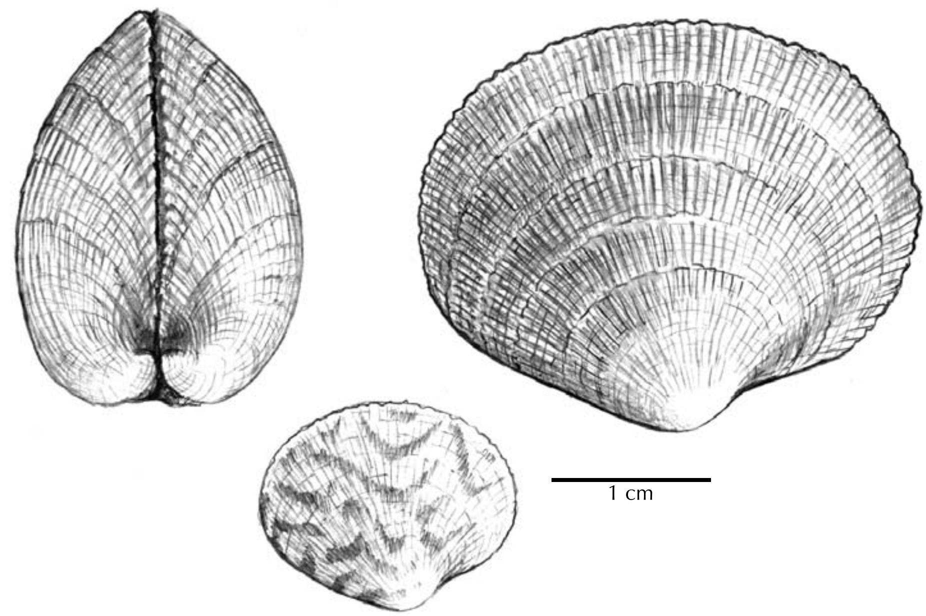

Plate 58. *Protothaca staminea*. McIntire Drawings, © 1999 by Zedler.

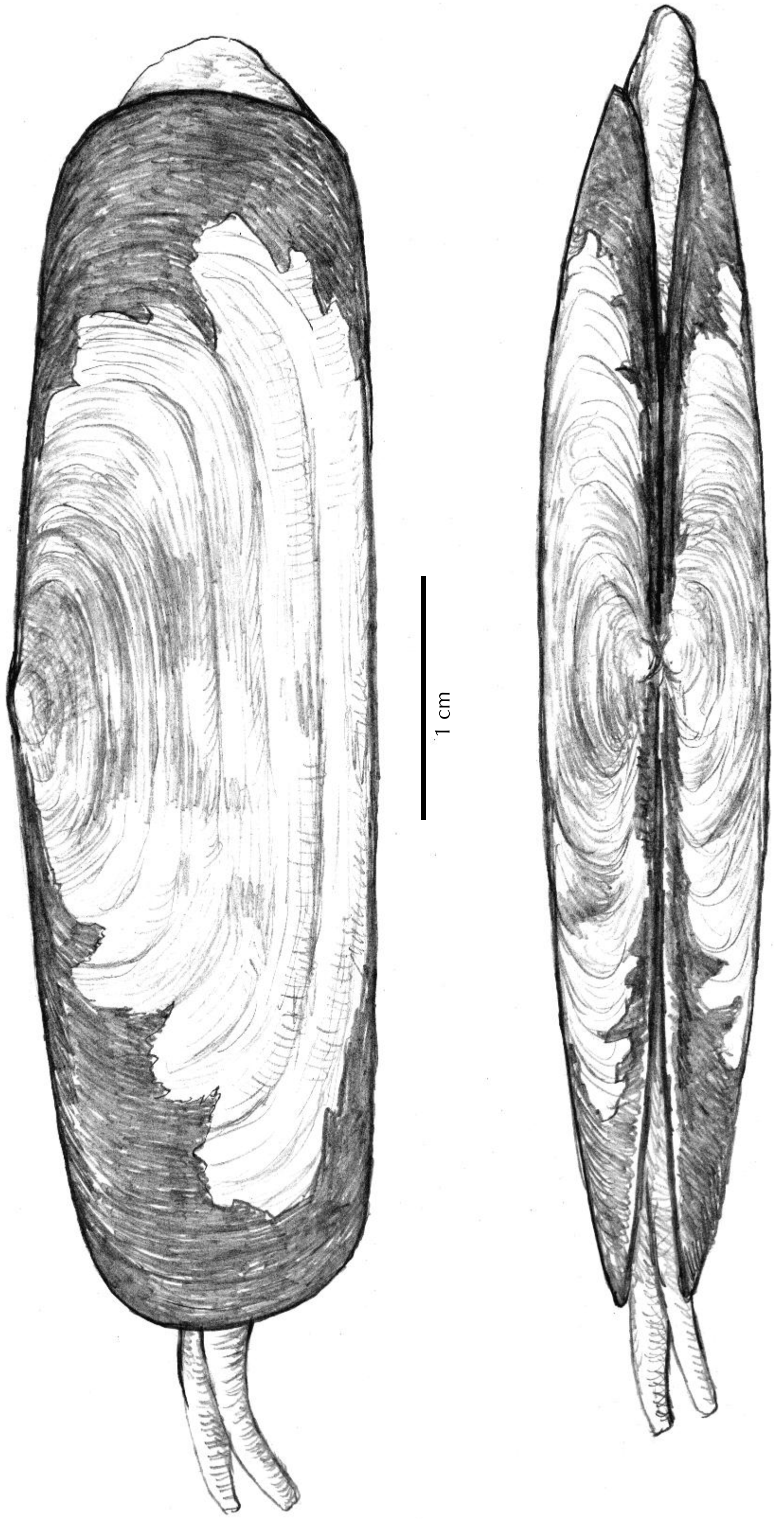

Plate 59. *Tagelus californianus.* McIntire Drawings, © 1999 by Zedler.

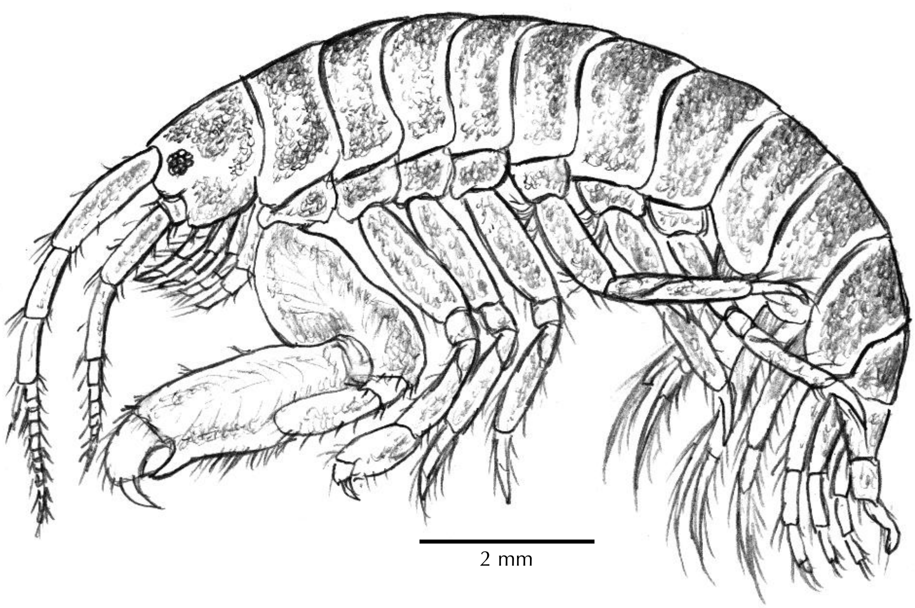

Plate 60. *Grandidierella japonica*. McIntire Drawings, © 1999 by Zedler.

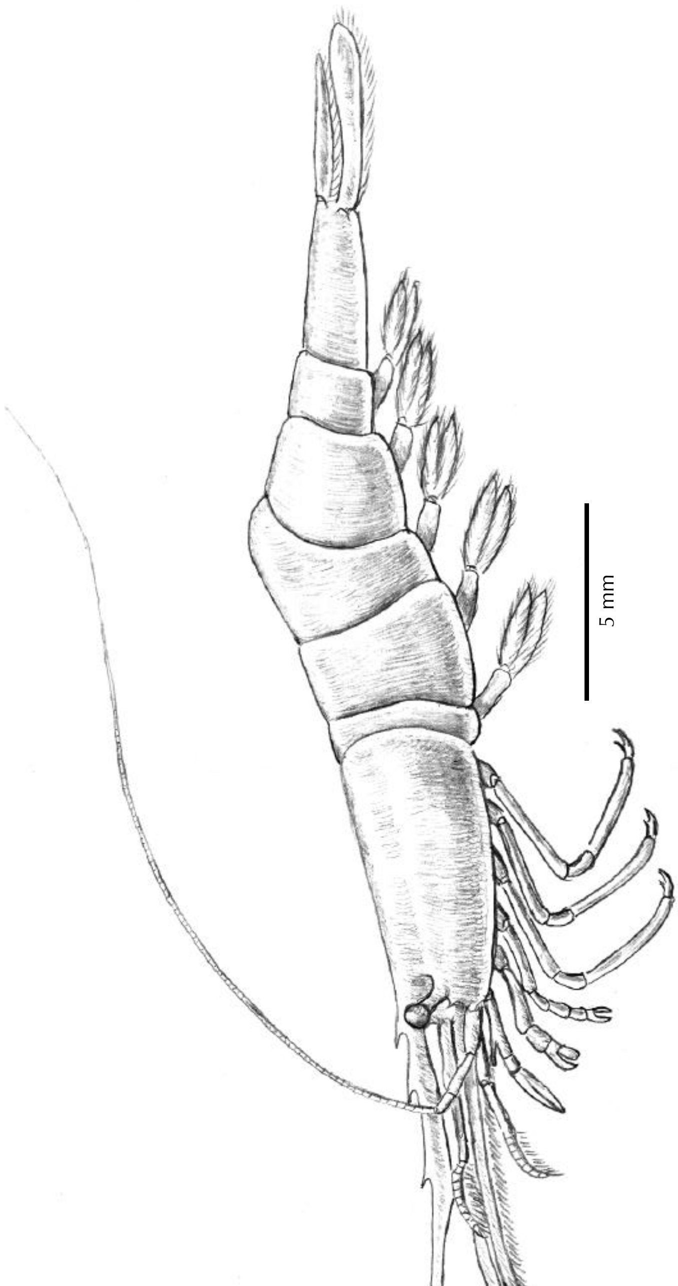

Plate 61. *Hippolyte californiensis*. McIntire Drawings, © 1999 by Zedler.

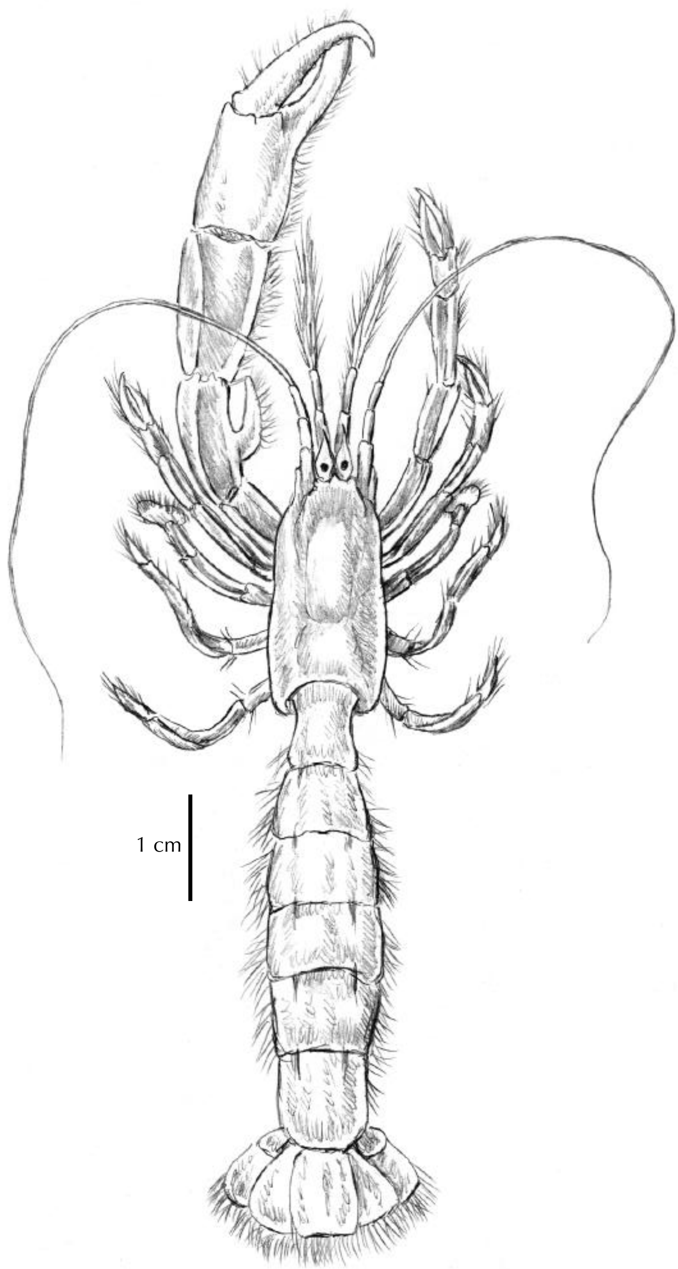

Plate 62. *Neotrypaea californiensis*. McIntire Drawings, © 1999 by Zedler.

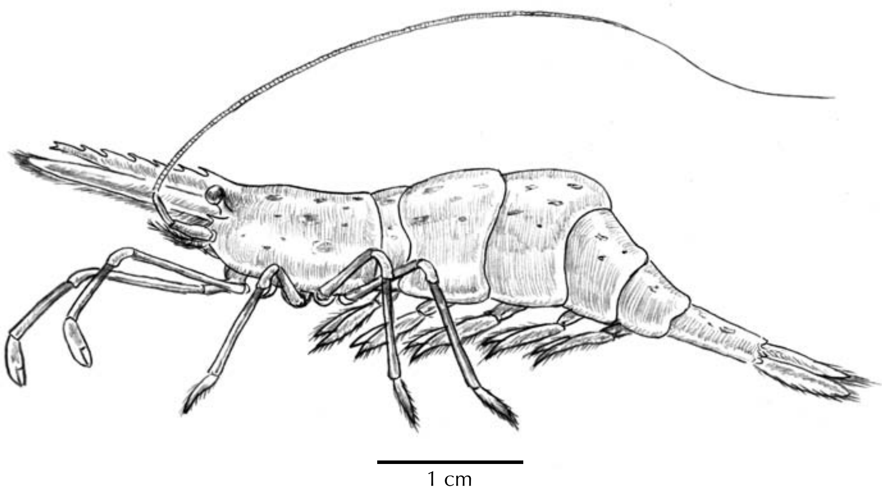

Plate 63. *Palaemon macrodactylus*. McIntire Drawings, © 1999 by Zedler.

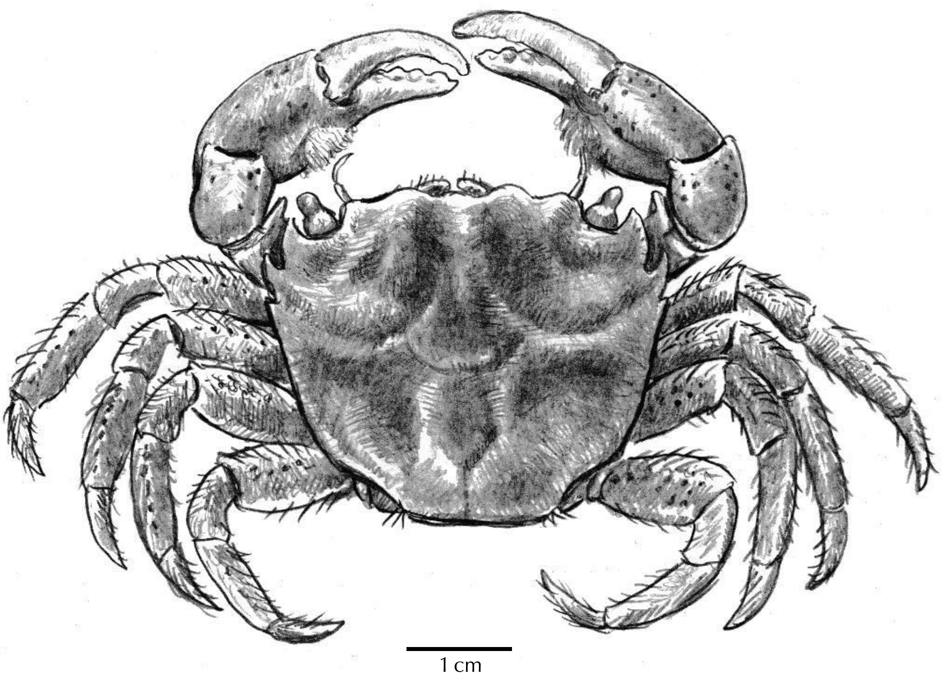

Plate 64. *Hemigrapsus oregonensis*. McIntire Drawings, © 1999 by Zedler.

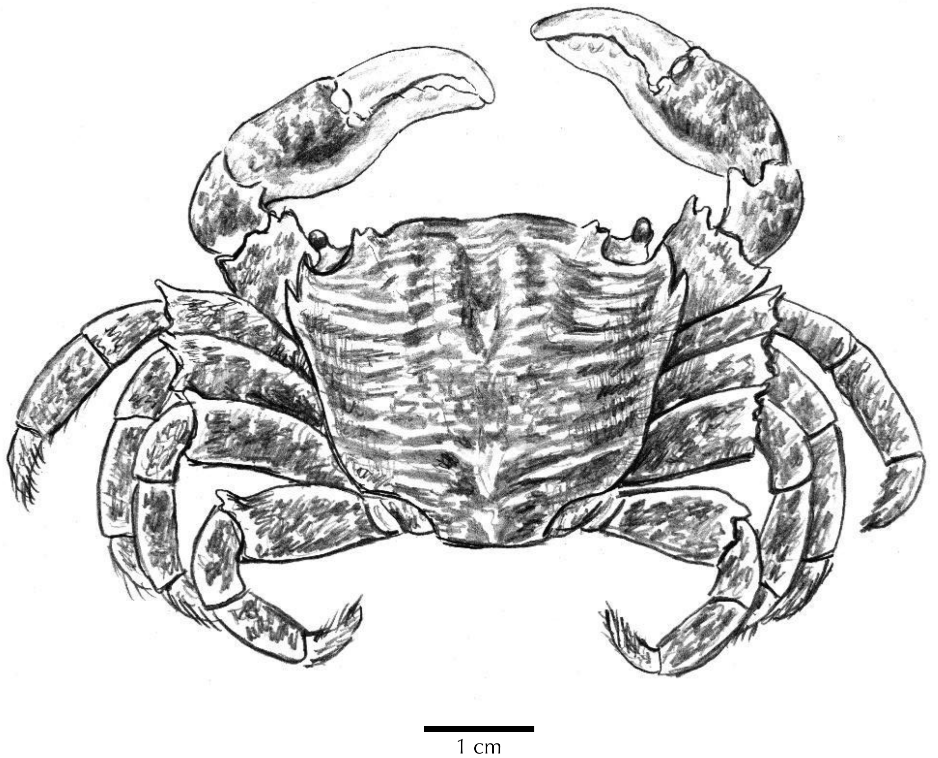

Plate 65. *Pachygrapsus crassipes*. McIntire Drawings, © 1999 by Zedler.

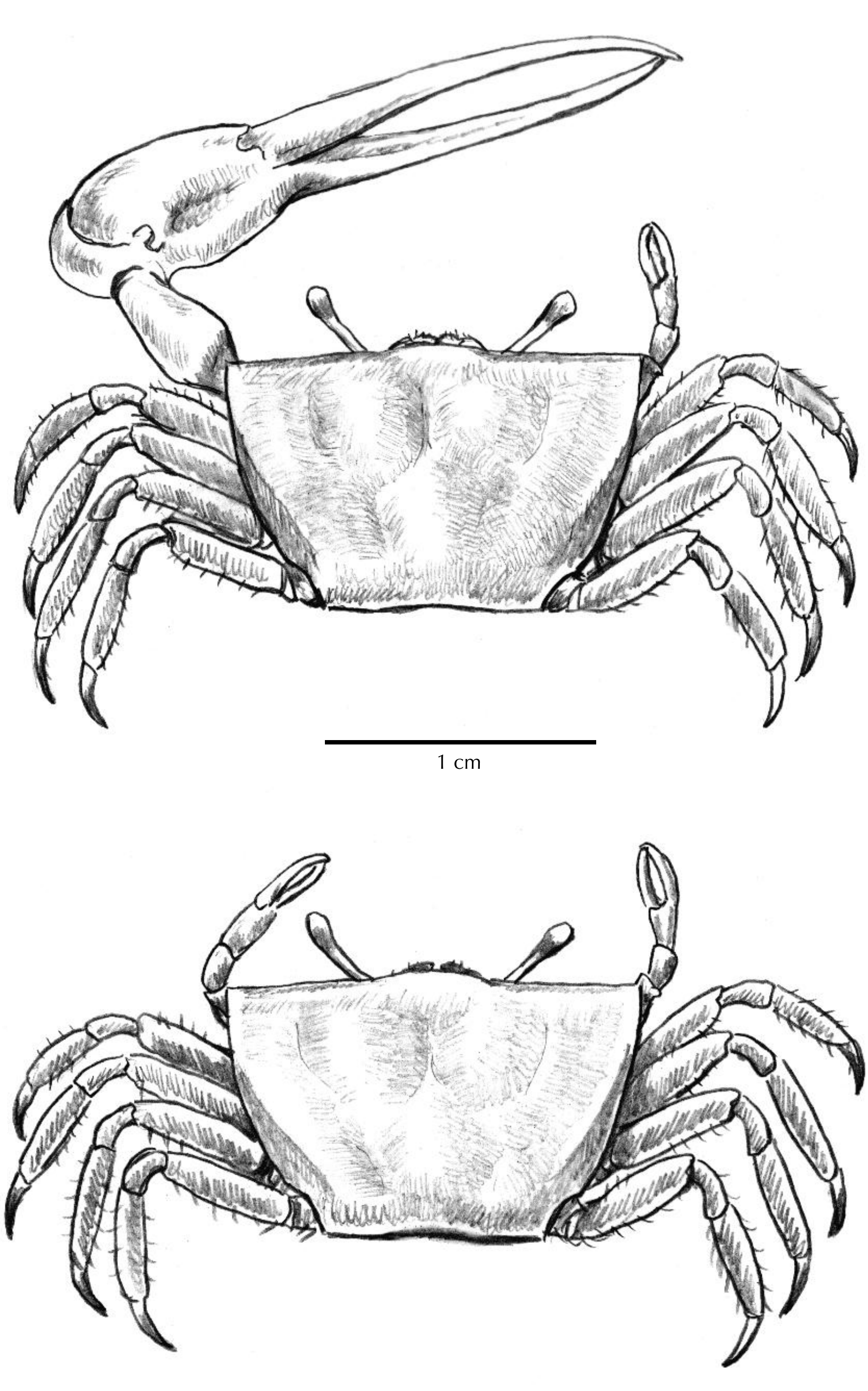

Plate 66. *Uca crenulata*. The top drawing represents a male; the bottom drawing shows a female. McIntire Drawings, © 1999 by Zedler.

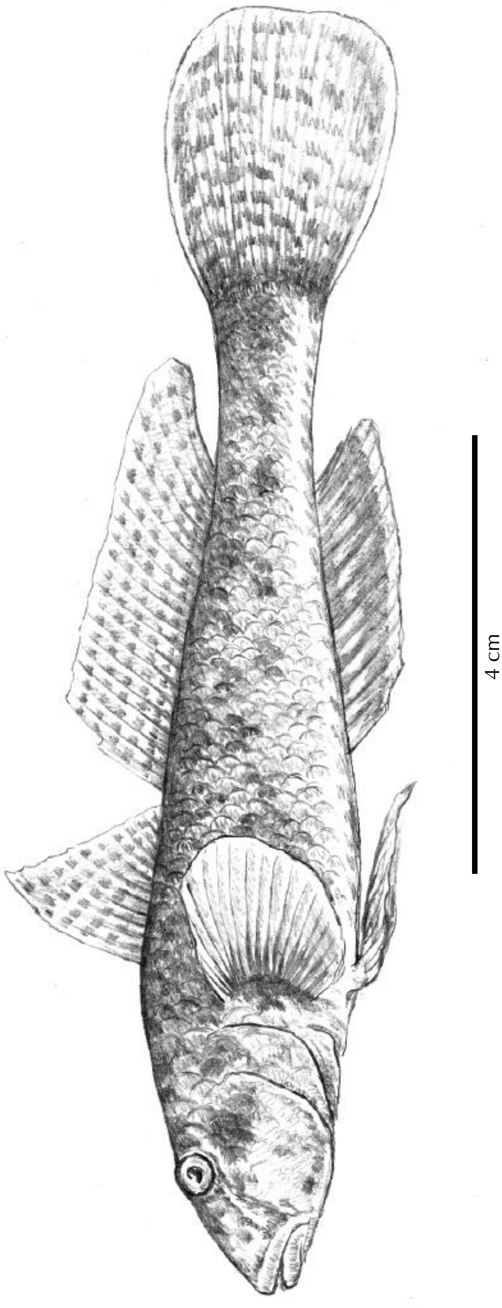

Plate 67. *Acanthogobius flavimanus*. McIntire Drawings, © 1999 by Zedler.

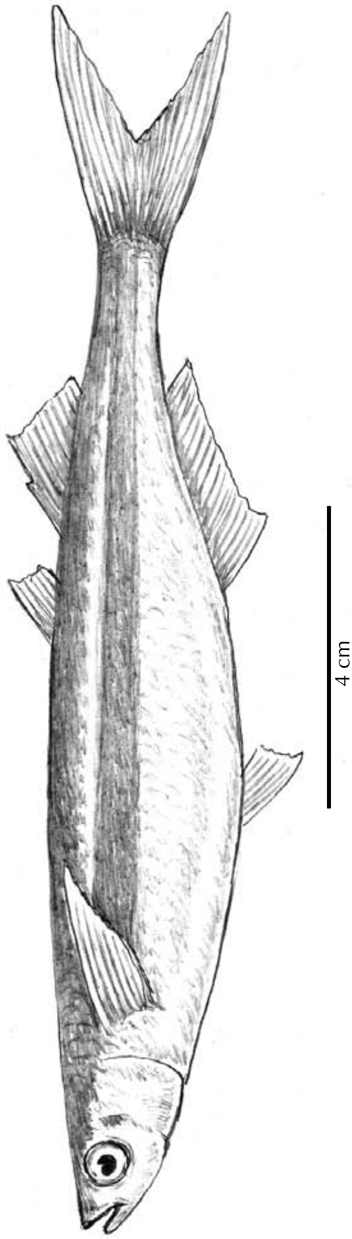

Plate 68. *Atherinops affinis*. McIntire Drawings, © 1999 by Zedler.

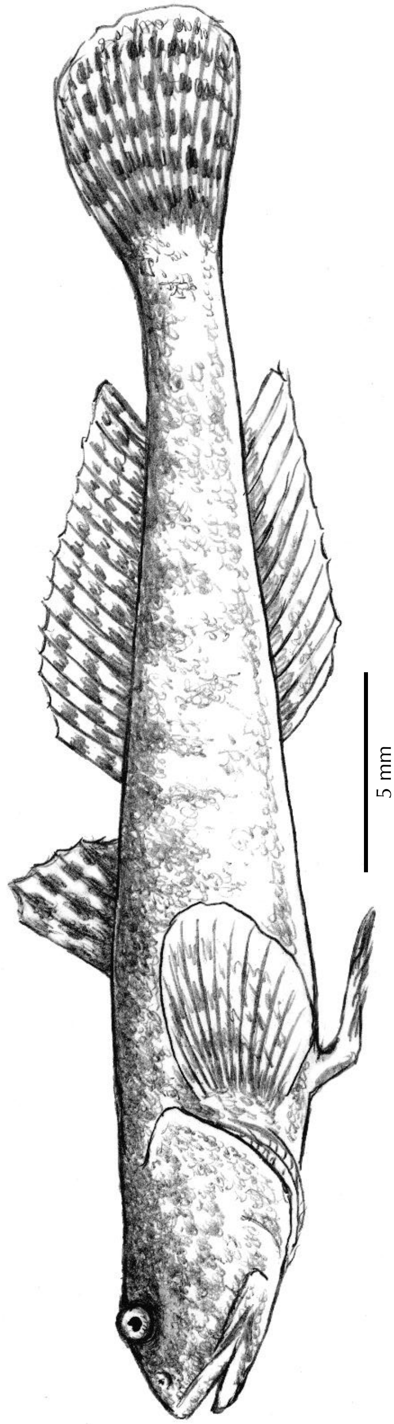

Plate 69. *Clevelandia ios*. McIntire Drawings, © 1999 by Zedler.

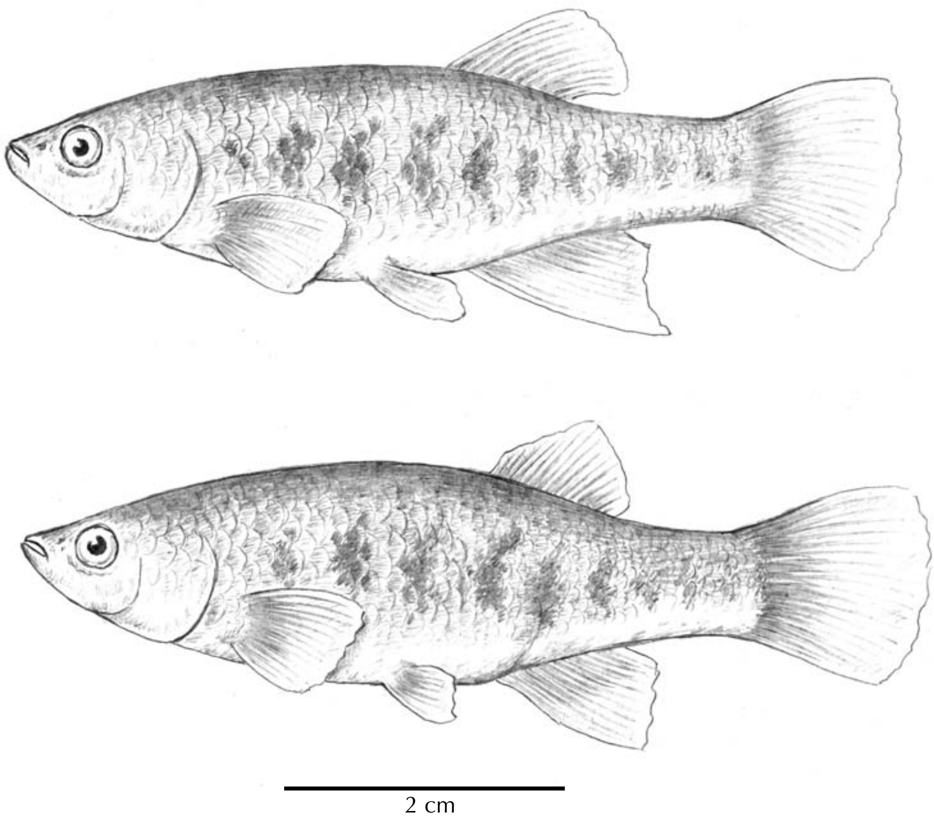

Plate 70. *Fundulus parvipinnis*. The top drawing represents a male; the bottom drawing shows a female. McIntire Drawings, © 1999 by Zedler.

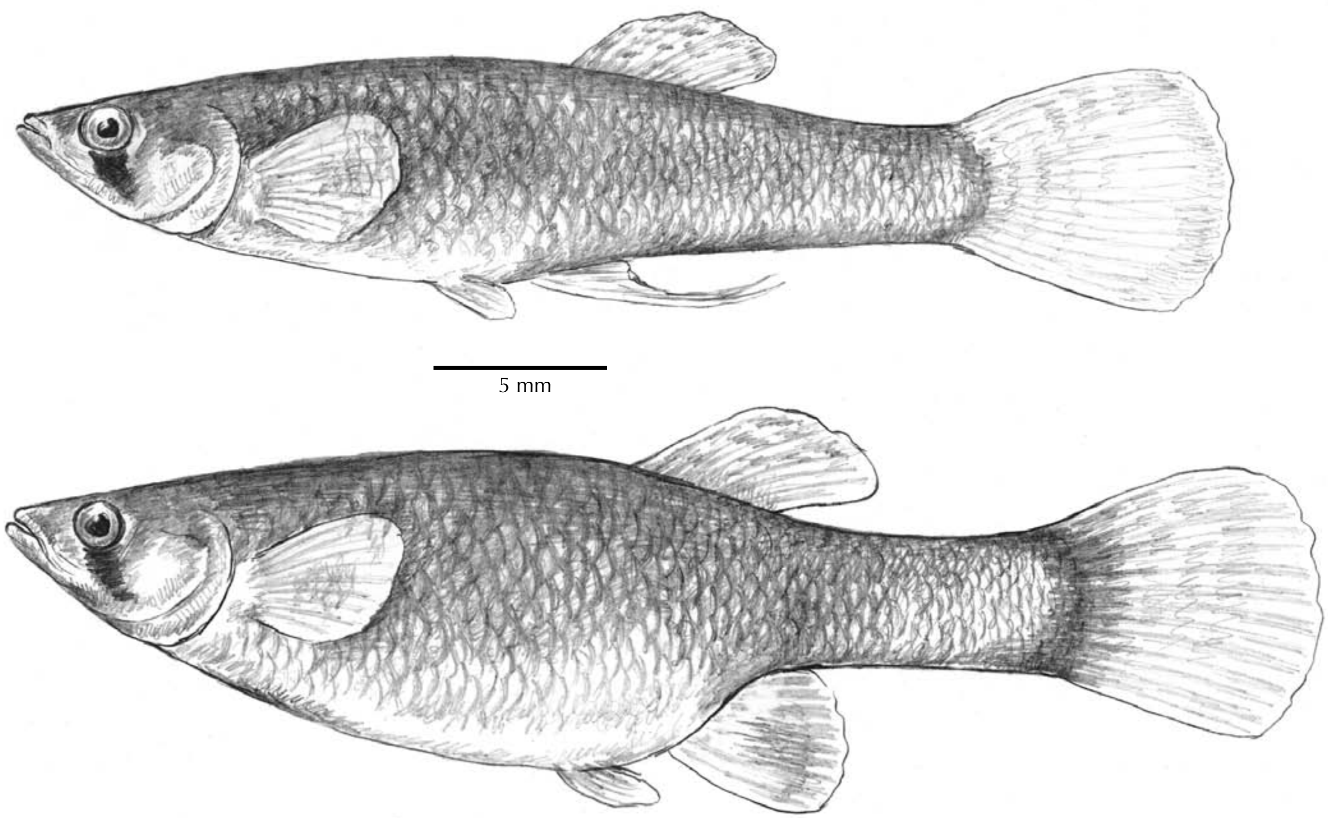

Plate 71. *Gambusia affinis*. The top drawing represents a male; the bottom drawing shows a female. McIntire Drawings, © 1999 by Zedler.

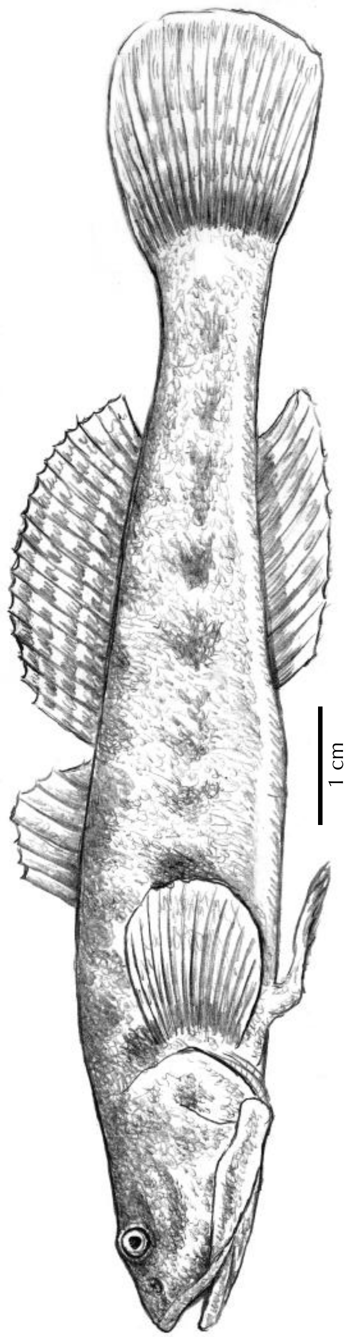

Plate 72. *Gillichthys mirabilis.* McIntire Drawings, © 1999 by Zedler.

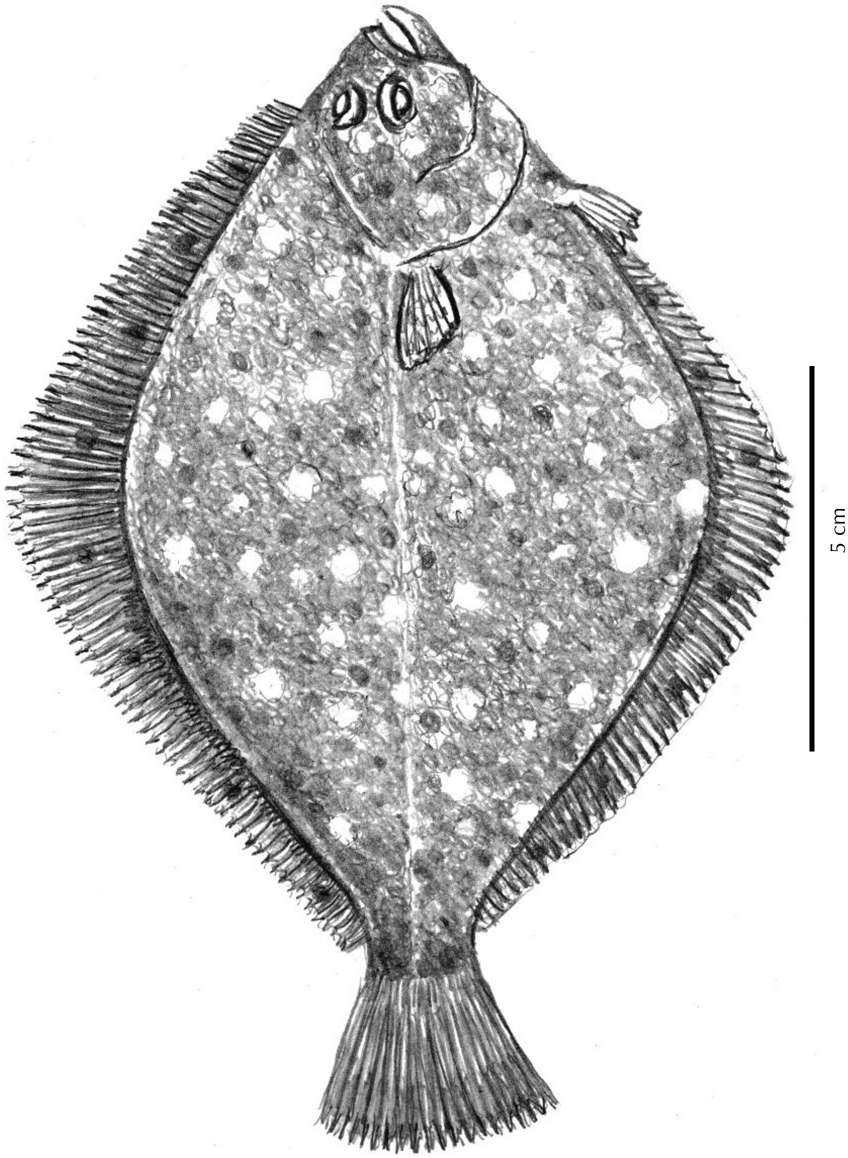

Plate 73. *Hypsopsetta gutulatta.* McIntire Drawings, © 1999 by Zedler.

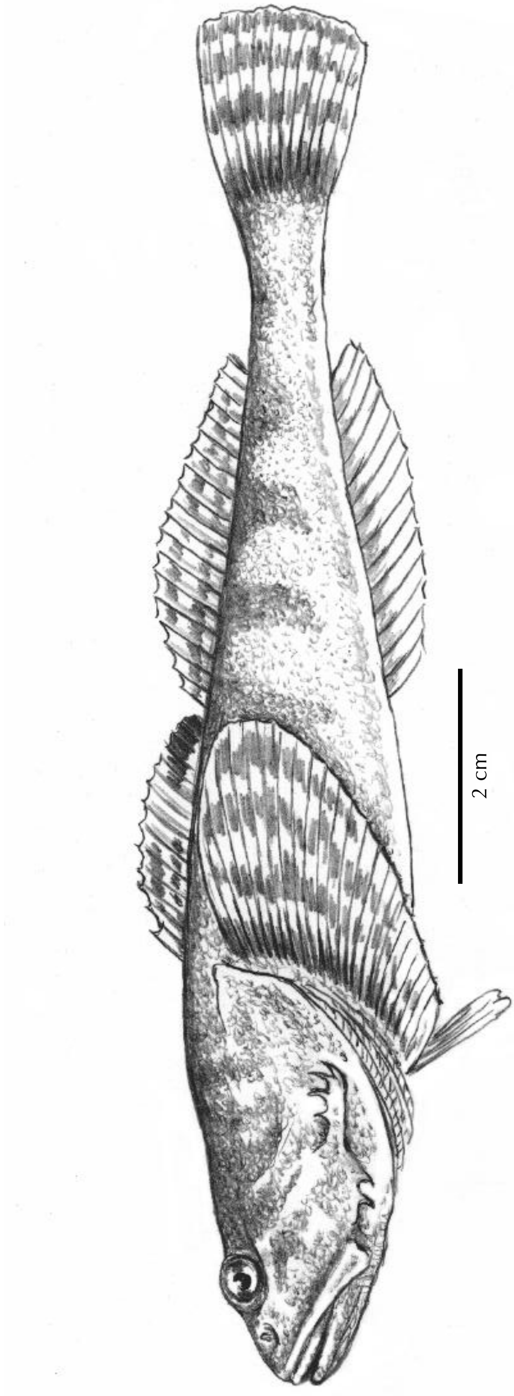

Plate 74. *Leptocottus armatus*. McIntire Drawings, © 1999 by Zedler.

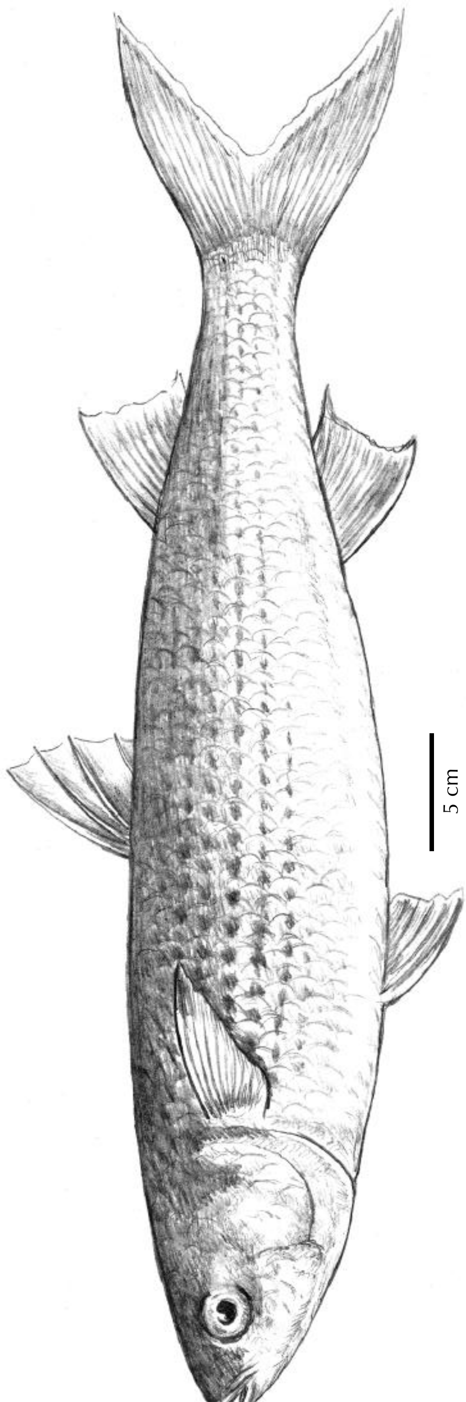

Plate 75. *Mugil cephalus.* McIntire Drawings, © 1999 by Zedler.

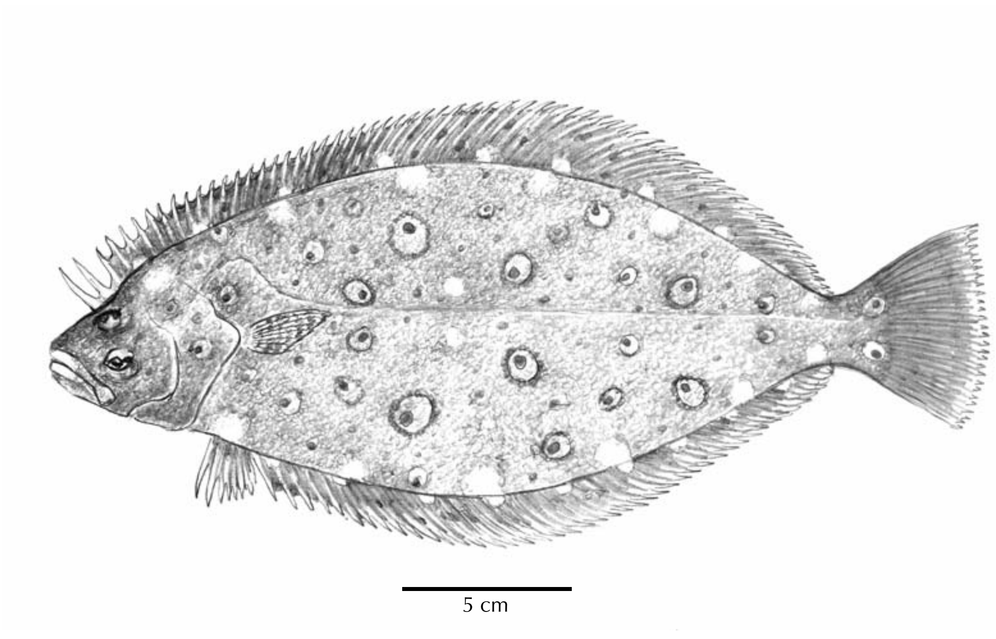

Plate 76. *Paralichthys californicus*. McIntire Drawings, © 1999 by Zedler.

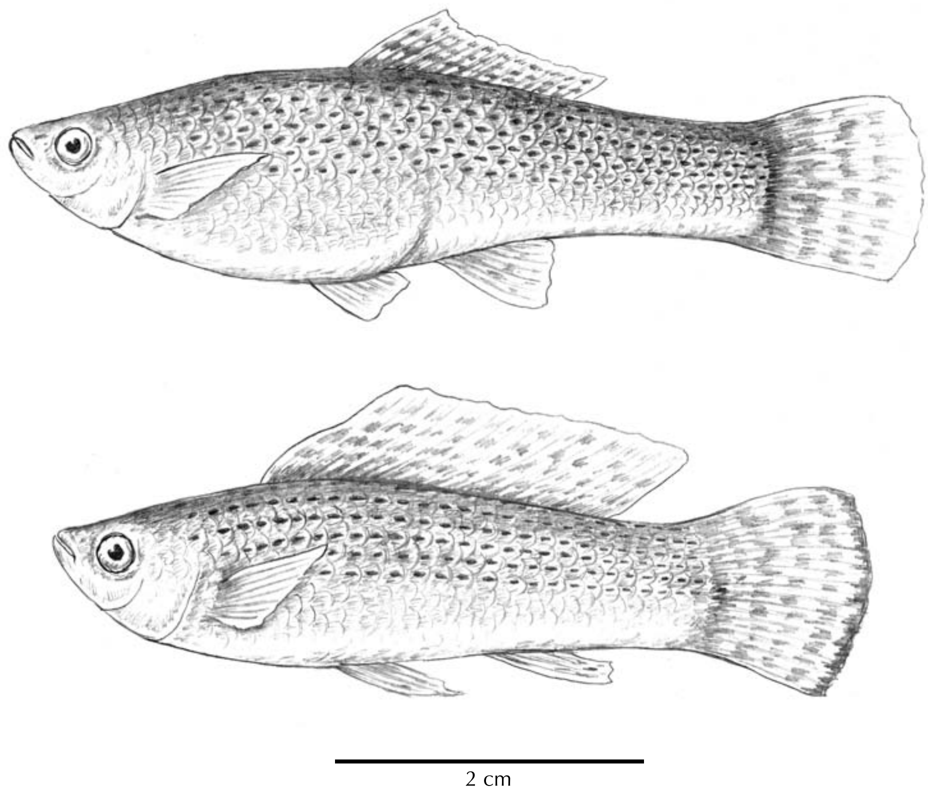

Plate 77. *Poecilia latipinna*. The top drawing represents a male; the bottom drawing shows a female. McIntire Drawings, © 1999 by Zedler.

chapter five

Restoring assemblages of invertebrates and fishes

Gregory D. Williams and Julie S. Desmond

5.1 Overview

Fish and invertebrates are integral and valued components of naturally functioning wetland ecosystems. Benthic invertebrates provide food-web support for consumers such as shorebirds and fish (Virnstein 1977, Schneider 1978, Quammen 1984) and influence sediment properties, including compaction (McMaster 1967), water content, and texture (Rhoads and Young 1970). Similarly, fish function as vehicles for nutrient cycling and energy transfer across habitats at a number of trophic levels in the estuarine food web (Allen 1982, Kneib 1997, Kwak and Zedler 1997). In turn, wetland habitats provide these animals with areas for refuge, reproduction, feeding, and other essential functions. For example, it is estimated that over 75% of the commercially important fish and invertebrates harvested annually in the U.S. rely upon coastal wetlands during some part of their life cycle (Chambers 1992). While many southern California wetland fish and invertebrate species are not considered commercially or recreationally important, they are vital links in the nearshore food web, serving as prey for endangered bird species, such as the California least tern (*Sterna antillarum browni*) and the light-footed clapper rail (*Rallus longirostris levipes*) (Zedler et al. 1992).

Because of the important roles fish and invertebrates play in wetland ecosystems, many restoration projects are designed to provide habitat for these groups. Typical restoration goals are to provide nursery habitat for commercially important species or food-chain support for higher trophic levels. It is often assumed that animals will recolonize a site naturally if the native plant community is present, and sites are designed to meet the needs of various plant assemblages (Chapter 4). Little attention is paid to the factors influencing animal colonization and persistence, including how humans affect the ability of native fish and invertebrates to use areas with altered water quality, introduced species, and night lighting.

Fish and most macrocrustaceans are highly mobile, and they rely on a variety of habitats over short time scales for different functions (Miller and Dunn 1980). In contrast, many benthic invertebrates are sessile. Some species lack a planktonic larval stage and have extremely limited dispersal throughout their lives (Levin 1984). Others have a planktonic larval stage and select habitat suitable for settling out of the plankton. Temperature, salinity, and habitat structure (vegetation, sediment type, etc.) are important determinants

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of the fish and invertebrate species that colonize a habitat, as well as their abundances. Biotic interactions (predation, competition) and linkages to other habitats (landscape position) have additional effects. In this chapter we provide a brief review of the environmental parameters that influence the ecology and composition of wetland fish and macroinvertebrate assemblages in both natural and restored sites (Section 5.2). We then describe how typical wetland habitats are used by fish and macroinvertebrates, with specific examples of restoration approaches (Section 5.3).

5.2 Environmental parameters

Hydrology and topography are responsible for most physical and chemical features of wetlands, including tidal inundation patterns, water depth, and current speeds (Chapter 3). These habitat features affect the degree of use by fish and invertebrates, including the pattern of distribution (McIvor and Rozas 1996), potential for colonization and growth, support of food chains, refuges from predation, reproduction, and nursery functions. The relative volumes of tidal and freshwater (riverine) flows further influence fish and invertebrate assemblages. Studies conducted in a variety of aquatic settings have shown that species-specific fish-habitat associations are related to flow velocity, depth, and stream order (Gorman and Karr 1978, Meffe and Sheldon 1988, Baltz et al. 1993, Ruiz et al. 1993, Paller 1994, Kirchhofer 1995). These hydrologic elements combine in marsh settings to form landscape features that we describe as stream order and subtidal geomorphology. Other habitat features, such as substrate type, vegetation type and cover, and physical structure are covariates of hydrology and elevation, and these factors are often inseparable.

As the foundation for marsh development and maturation, hydrology should form the basis for any restoration plan (Zedler 1996*a*,*b*). In this section we use the ecological literature from both natural and restored systems to describe how habitat features influence fish and invertebrate assemblages. The processes that control fish and invertebrate distributions in natural systems appear to be the same in constructed systems. We summarize weaknesses in previous restoration projects and make suggestions on how to correct such problems in future plans.

5.2.1 Elevation

Elevation and the tidal cycle interact to determine water depth and submergence time in marsh habitats. Benthic invertebrate species display a distinctive vertical distribution pattern along limits of physiological stress (e.g., desiccation, temperature shock, osmotic imbalance, cessation of feeding and aerobic respiration) (Peterson 1991). For example, polychaete and total invertebrate densities have been shown to decrease with increasing elevation and distance from the nearest water source in *Spartina alterniflora* and *Juncus* marshes in Galveston Bay, Texas (Minello et al. 1994), Sapelo Island, Georgia (Kneib 1984), north Florida (Subrahmanyam and Coultas 1976), and North Carolina (Moy and Levin 1991).

The elevation of a marsh can influence the rate at which invertebrates colonize by limiting the duration of tidal inundation and associated exposure time for planktonic larval recruitment (Eckman 1983). In constructed *Spartina alterniflora* marshes in North Carolina, infaunal communities appeared to develop more rapidly in the low marsh than the high marsh, although the habitats differed by only 20 cm (Moy and Levin 1991, Levin et al. 1996). Along the Gulf Coast of Texas, fish and crustacean densities were significantly lower in created *S. alterniflora* salt marshes, where tidal flooding regimes were more variable and elevations higher than in natural marshes (Minello and Webb 1997). Elevation

can also influence fish use of salt marshes. In Humboldt Bay, California, fish use was lower in a created *Salicornia virginica/Spartina foliosa* marsh than in a nearby natural marsh, which had a lower elevation (Chamberlain and Barnhart 1993).

Tied to elevation are biotic factors such as competition and predation, which have been implicated as causes of intertidal zonation in benthic communities (Peterson 1979). Elevation can influence the distribution of predators, and this in turn can dictate the distribution of prey species. For example, Posey (1986) showed that the lower distributional limit of burrowing shrimp (*Callianassa californiensis*) in intertidal sandflats was influenced by staghorn sculpin (*Leptocottus armatus*), a common predatory fish. Likewise, the California horn snail (*Cerithidea californica*) occurred most commonly in salt marsh pannes above the desiccation tolerance of its predator, the dog whelk (*Ilyanassa obsoleta*) (Race 1982).

Water depth has a demonstrated role as a refuge. In non-vegetated estuarine habitats in Chesapeake Bay, small fish, such as mummichog (*Fundulus heteroclitus*), and epibenthic crustaceans, such as grass shrimp and blue crabs (*Palaemonetes pugio* and *Callinectes sapidus*), were most abundant in shallow waters; depth-related changes in size distribution were correlated with changes in predation risk (Ruiz et al. 1993). In vegetated marshes, snails and crabs were at less risk to predation in shallow habitats (Hamilton 1976, Wilson 1989).

Marshes constructed with mostly low elevation marsh habitat may develop infaunal invertebrate communities more quickly and have higher densities and different species assemblages of nekton than those with higher elevations. Where shallow water edge habitats are minimal, however, small species may lack adequate refuge from large subtidal predators (e.g., California halibut). Including both deep- and shallow-water refuges can increase the functional diversity for a wider variety of animal species.

5.2.2 Flow rates

Distributions of fish and invertebrates are mediated by flow rate (current velocity) because many aquatic organisms rely on the currents to deliver food, carry away metabolic wastes, and facilitate gas exchange. Suspension-feeding invertebrates, which filter food items from the water column, are often incapable of living in areas with low flow rates, likely because the sestonic food supply is lower there (Sanders 1958). Flow rates can also influence the feeding rates of deposit-feeders, which process sediments to collect food (Levinton 1971). Some spionid polychaetes, for instance, may even alter their feeding mode by shifting from surface deposit-feeding to suspension-feeding if flow rates and particulate fluxes are high (Taghon et al. 1980).

Flow may also determine distribution patterns by influencing both larval settlement (Eckman 1983) and habitat selection, based on the size and hydrodynamic characteristics of the fish or invertebrate (Howard and Nunny 1983, Maude and Williams 1983). Baltz et al. (1993) observed that some species (e.g., juvenile blackcheek tonguefish, *Symphurus plagiusa*) in Louisiana estuaries shift to higher velocity microhabitats with increasing size. Finally, flow can influence an organism's interactions with predators or competitors by altering predation efficiency (Hansen et al. 1991, Hart 1992) or chemosensory abilities (Weissburg and Zimmer-Faust 1993).

In freshwater streams, flow rate is one of the most important habitat variables affecting fish distributions (Gorman and Karr 1978, Moyle and Baltz 1985, Bain et al. 1988, Meffe and Sheldon 1988). As a result, most analyses of perturbation and restoration projects in streams pay special attention to landscape features and habitat structural characteristics. For example, large woody debris influences flow rates, the formation of refugia, and

habitat heterogeneity (Wesche 1985). Stream flow rate has also been acknowledged as critical for restoring habitat for freshwater macroinvertebrates (Gore 1985). Habitat-enhancing structures (trees, boulders) that alter velocity and depth and provide suitable islands for macroinvertebrate recolonization are often used in the restoration of stream channels. Although flow rate needs to be considered in the restoration of salt marsh habitats for fish and invertebrates, the requirements and tolerances of different species need further study.

5.2.3 Substrate

Substrate texture is an important determinant of invertebrate distributions (Sanders 1958, Rhoads and Young 1970, Levinton 1972). Suspension-feeders are often found in large-grained, sandy sediments, where currents are generally greater, where the planktonic food supply is more frequently replenished (Peterson 1991), and where the activities of deposit-feeding organisms are less likely to interfere with filtering and larval settlement (Rhoads and Young 1970). Marsh substrate texture is determined by flow rate through the selective deposition of sediment particles of different grain size. Low flows allow fine particles and organic matter to settle out, resulting in dominance by silt and clay. High flow areas are dominated by coarse sands (Chapter 3). Fine-grained sediments have high organic content and are usually dominated by deposit-feeding organisms such as polychaetes, oligochaetes, or amphipods.

Most natural marshes are characterized by fine soils containing relatively large amounts of organic matter. Dense communities of subsurface deposit-feeding organisms, such as tubificid and enchytraeid oligochaetes (Moy and Levin 1991, Levin et al. 1996, Levin et al. 1998), either feed on organic matter directly or consume the associated bacteria (Lopez and Levinton 1987). Marshes constructed from coarse dredge spoil material contain less organic matter and, presumably, less food for invertebrates. The coarse soils of constructed marshes may also be inappropriate for the construction of tubes and burrows. Coarse substrate may limit the capacity of constructed marshes to support the communities found in reference marshes. Constructed marshes often support fewer subsurface deposit-feeders than natural marshes (Moy and Levin 1991, Levin et al. 1996), and they are often dominated by surface-deposit feeders such as *Streblospio benedicti* (Moy and Levin 1991, Levin et al. 1996, MEC Analytic Systems Inc. 1995).

Although soil amendments (organic matter and fertilizers) are used to promote plant growth at constructed sites (Valiela et al. 1975, 1976, Gibson et al. 1994, Boyer and Zedler 1996), they are less often considered as a method to promote invertebrate colonization. In the Great Sippiwissett Marsh in Massachusetts, Milorganite® (a sewage-derived amendment) was added to a tidal creek for 15 years (Sarda et al. 1996). Over this period, the treated creek switched from dominance by the polychaete *Streblospio benedicti* to dominance by subsurface deposit feeding oligochaetes, while a nearby control creek remained dominated by *S. benedicti*. Results of other studies support the idea that soil organic matter is important for oligochaetes (Moy and Levin 1991, Levin et al. 1996). However, in some cases, adding organic matter may lead to anoxic conditions. This was the case in a *Spartina alterniflora* marsh in North Carolina, where amendments initially slowed the colonization of plots by macrofauna (Levin et al. 1997*a*). Because soil amendments can speed the development of dense infaunal communities where sediments are low in organic matter, there is potential for negative effects, and on-site trials may be needed.

Sediment properties also affect fish activities, including feeding. Along the Atlantic Coast, juvenile spot (*Leiostomus xanthurus*) demonstrate a clear preference for feeding on meiofauna in fine-textured substrate because prey are concentrated, accessible, and easily processed in mud (Smith and Coull 1987). For flatfish, sediment size influences burial

ability, an important behavior that allows fish to escape predation and avoid being seen by prey. In juvenile plaice (*Pleuronectes platessa*), burial ability in large-grained sediments was found to increase with body size (Gibson and Robb 1992). Smaller fish could bury in finer sediments, but larger fish could bury in larger-grained sediments. Thus, sediment grain size may be an important consideration in restoring habitats for some fishes.

5.2.4 Vegetation

Water depth, submergence time, and flow rates directly influence the composition and density of salt marsh vegetation (Adam 1990). The vegetation affects aquatic community composition (Heck and Crowder 1991) by providing a refuge for fish and invertebrates and influencing predator-prey interactions (Vince et al. 1976, Rozas and Odum 1988). Fish and invertebrate richness and biomass are usually many times greater in or near vegetated than unvegetated habitats. This is true for seagrass beds (Stoner 1980, Heck and Thoman 1984, Irlandi and Crawford 1997), mangroves (Robertson and Duke 1987), tidal salt marshes (Zimmerman and Minello 1984, Peterson and Turner 1994), and tidal freshwater marshes (Rozas and Odum 1988). The patterns have generally been attributed to the combination of available foods and predator protection. It is thus of some concern that plant densities are often lower in restored habitats than in reference sites.

The relationship between vegetation and infauna seems to depend on the vegetation type and various predator-prey attributes, including mobility and size (Orth et al. 1984, Ryer 1988, Lana and Guiss 1992). Root masses may create protective refugia (Blundon and Kennedy 1982, Peterson 1982, Lana and Guiss 1992) or enhance food supplies for some taxa, while aboveground growth can potentially alter larval settlement via flow patterns or behavior (i.e., larval habitat selection, Rader 1984). While some studies of infauna and vegetation reveal little pattern, others show a relationship. Lana and Guiss (1992) found that infaunal abundance was associated with belowground biomass in *Spartina alterniflora* habitats in southeast Brazil, with roots used principally as refugia or structural support for their tubes. The species benefiting primarily from the presence of vegetation were two polychaetes, *Isolda pulchella* and *Nereis oligohalina*. Similarly, Rader (1984) found densities of infauna (all taxa collected, except for *Nereis succinea*) to be higher in sediment cores that contained *Spartina alterniflora* plugs than in unvegetated cores. In contrast, Capehart and Hackney (1989) found the density of clams (*Polymesoda carolinia*) to be negatively correlated with root density in a South Carolina marsh.

In addition to plant density, vegetation type has also been shown to influence the infaunal community. In North Carolina, Posey et al. (1997) found that the infaunal community tracked the development of vegetation at a restored site. A greater abundance of certain polychaetes appeared in newer (7-year-old) sites dominated by *Spartina alterniflora*, while higher densities of particular oligochaetes and amphipods occurred in the older (11 or 15-year-old) sites dominated by *Schoenoplectus robustus*. In a southern California marsh, polychaetes were denser in sediments vegetated by *Spartina foliosa*, while gastropods, isopods, and tubificid oligochaetes were more abundant in sediments vegetated by *Salicornia virginica* (Levin et al. 1997*b*). The type of vegetation planted or allowed to colonize a restored site can clearly influence the developing invertebrate community, although the consequences to marsh functioning are not fully explored.

Vegetation influences the abundance of nekton, which includes both fish and swimming crustaceans. Moy and Levin (1991) found that mummichog (*Fundulus heteroclitus*) abundance was lower in 3-year-old planted marshes than in natural marshes in North Carolina. They attributed this correlation to the lower *Spartina alterniflora* densities in constructed marshes, which may not have provided the fish with sufficient spawning sites and predation refugia. Similarly, in Virginia, Havens et al. (1995) found lower densities of

blue crab *Callinectes sapidus* in a 5-year-old constructed marsh relative to nearby natural marshes, which had denser *Spartina*. Rozas and Odum (1988) suggested that submerged plant beds in tidal freshwater creeks supported large concentrations of fishes because they enhanced foraging profitability and afforded protection from predators. However, as with invertebrates, the relationship between vegetation density and fish use may vary with species. In Texas, 2- to 5-year-old transplanted *Spartina alterniflora* marshes had higher plant densities than nearby natural marshes, but lower densities of decapod crustaceans (Minello and Zimmerman 1992).

Seasonal algal blooms can also produce discrete patterns of infaunal distribution through anoxia-induced mortality (Nichols and Thompson 1985, Everett 1991). Floating and entrained mats of macroalgae often characterize wetland habitats constructed in areas with poor tidal circulation or eutrophic conditions (Section 5.3.3). However, macroalgae add structural complexity to otherwise unvegetated mudflats or creeks, providing important habitat for some epibenthic invertebrates (Everett 1994) and fishes (Sogard and Able 1991).

5.2.5 Stream order and subtidal morphology

Stream order and subtidal geomorphology are habitat features that correlate with patterns of nektonic species composition and abundance in salt marsh habitats (McIvor and Rozas 1996, Williams and Zedler 1999). These features are the products of the marsh landscape and channel hydrology and are further correlated to a number of individual habitat parameters such as flow, substrate, and bank slope.

Hydrology, geology, and landscape features modify the number and hierarchy of tributaries flowing into a tidal channel. Tidal channels are classified by "stream order" (Horton 1945, Strahler 1964) as follows: A first-order channel has no tributaries; a second-order channel is formed when two first-order channels join; a third-order channel is formed when two second-order channels join, etc. A low-order channel feeding into a higher-order channel does not increase that channel's order (Figure 5.1). Predictably, channel order is correlated with a number of other physical habitat characteristics, including presence of vegetation (Rozas and Odum 1987*a*), temperature, salinity (Hackney et al. 1976), and depth, all of which can modify and influence habitat functions.

Rozas and Odum (1987*a*) tested the hypothesis that stream order influences species abundance patterns in tidal freshwater marshes of Virginia and found that fish numbers were greater in small headwater and main creek marshes than in larger river marshes. They attributed this pattern primarily to the increased abundance of refuges provided by submerged aquatic vegetation in the lower-order creeks. Hettler (1989) confirmed this pattern in tidal salt marshes in North Carolina, finding significantly more individuals and higher biomass (but lower diversity) of nekton in marshes adjacent to first-order creeks than in those adjacent to third-order channels. Several of the dominant species by numbers and biomass were predominantly collected from rivulet marsh; the dominants included mummichog (*Fundulus heteroclitus*), sheepshead minnow (*Cyprinodon variegatus*), striped killifish (*Fundulus majalis*), white mullet (*Mugil curema*), and spotfin mojarra (*Eucinostomus argenteus*) (Figure 5.2). In the Sacramento-San Joaquin Estuary in California, species composition varied distinctly by stream order. Small, first-order "sloughs" were mainly occupied by native resident species, while larger, higher-order channels were occupied by seasonal visitors (Moyle et al. 1986; Meng et al. 1994). In southern California, Desmond et al. (2000) found that size structure of juvenile California killifish (*Fundulus parvipinnis*) was strongly related to channel depth in a San Diego Bay marsh. Small juveniles seemed to have an affinity for shallow habitats, including first-order creeks and shallow (<0.25 m deep) areas in higher-order channels. All of these studies support the idea that small creeks

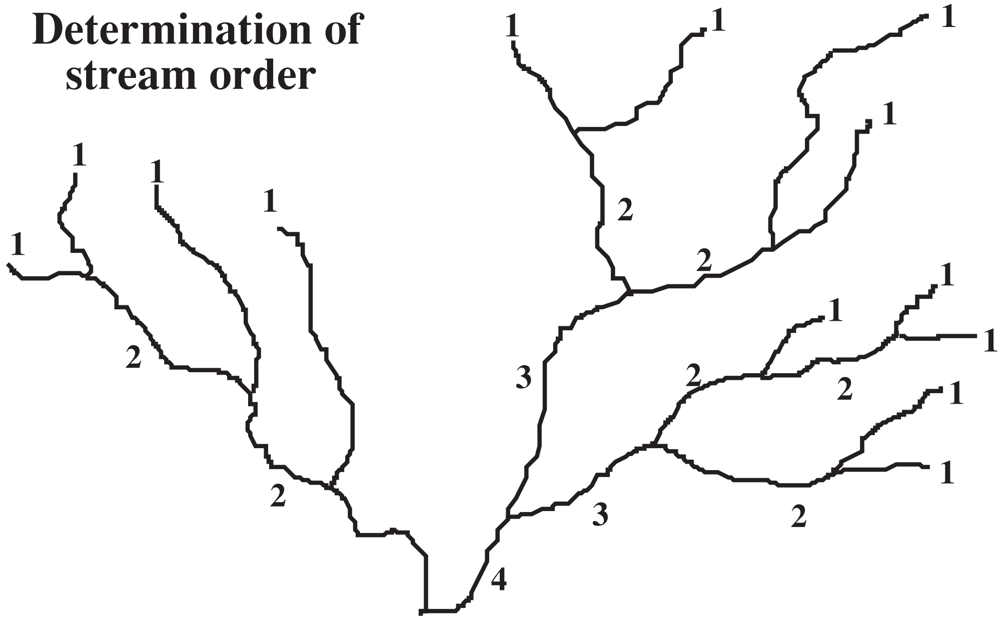

Figure 5.1 Derivation of stream order based on Horton's (1945) system. A first-order creek has no tributaries; a second-order creek is formed when two first-order creeks join; a third-order creek is formed when two second-order creeks join, and so on.

are important to resident marsh species. Thus, marshes with a diversity of hydrogeomorphic features, including complex creek networks, should support a greater range of ecosystem functions for fish than will more homogeneous habitats.

Marsh channel morphology is interrelated with a number of factors, including marsh sediment composition, vegetation, landscape features, and tidal energy (Garofalo 1980, Coats et al. 1995, Eisma et al. 1997). In one of the only marsh studies to investigate the role of subtidal geomorphology in mediating fish community patterns, McIvor and Odum (1988) demonstrated that tidal marshes adjacent to gradually sloping, depositional creek banks had higher fish densities than those adjacent to steep, erosional banks. They suggested that channel slope affected prey availability, predator encounter rates, and bioenergetic costs. In two California marshes, creek slope had the opposite effect on invasion by an exotic isopod, *Sphaeroma quoyanum*, which was much more concentrated in steeply sloped than gently sloped banks (Levin et al. 1999).

Williams and Zedler (1999) compared natural and constructed channel habitats along San Diego Bay, California, and integrated measures of channel stream order and morphology into their analysis of restoration efforts. Total fish abundance and species diversity did not differ between natural and constructed sites. However, analyses of individual species showed that subtidal morphology and hydrology explained most of the variability in fish assemblage composition (Figure 5.3). Correlations between restoration history and physical features were more an artifact of site choice than a cause-effect relationship. The constructed channels were primarily located in high-flow areas designed to link remnant marsh habitats and accommodate feeding by an endangered tern; hence they were primarily broad and deep, with sandy sediments and broad, gradually sloping banks. California killifish (*Fundulus parvipinnis*), a common species in vegetated marsh edge habitats, were found in significantly higher densities in the broad, shallow margins of these habitats. Selection of natural reference sites was constrained by the lack of relatively unaltered, accessible habitats, which were primarily represented by shallow, low-order channels with

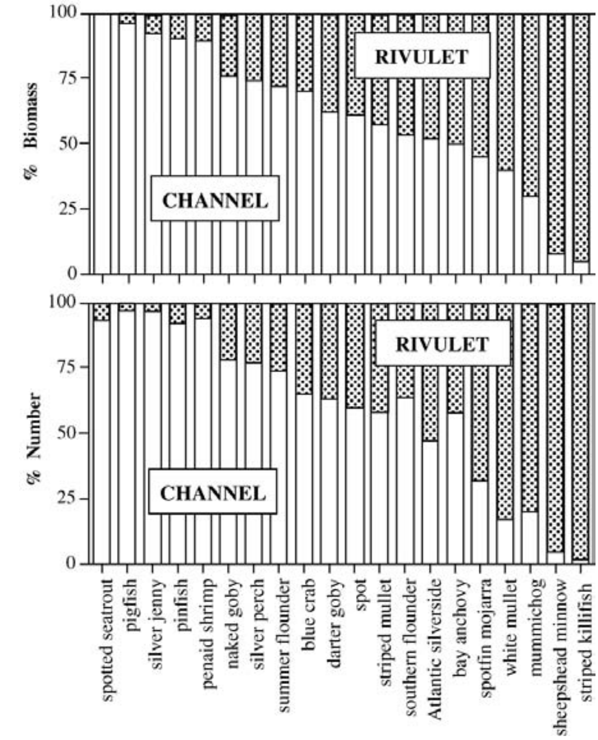

Figure 5.2 Percent number and percent biomass of fish species collected from block net samples on two types of *Spartina alterniflora* marsh near Beaufort, North Carolina, USA. "Rivulet" and "channel" marshes were adjacent to first- and third-order creeks, respectively. (Modified from Hettler 1989, Nekton use of regularly-flooded saltmarsh cordgrass habitat in North Carolina, USA. *Marine Ecology Progress Series* **56**:111-118. With permission.)

steep banks and muted tidal flushing. Longjaw mudsuckers (*Gillichthys mirabilis*), which tolerate low levels of dissolved oxygen, predominated in these habitats. Topsmelt (*Atherinops affinis*), a water-column species that is generally associated with subtidal habitats, dominated the large and deep channels at both natural and constructed sites. In future projects, marsh channel design should more closely match natural features (networks with several orders of creeks), and the assessment of channel biota in natural and constructed sites should be based on comparisons among creeks of similar hydrologic regimes and morphometry.

5.2.6 Landscape issues

Habitat heterogeneity, connectivity, and edge are all relevant design concepts for conserving biodiversity. Recent reviews of wetland habitat mitigation in southern California conclude that restoration sites need to incorporate greater complexity by increasing topographic heterogeneity and habitat linkages at multiple scales (Zedler 1996*b*, Zedler et al. 1997, Chapter 2). The argument that fishes and invertebrates require heterogeneous, connected landscapes is particularly strong.

Although many sessile estuarine invertebrates lack a planktonic stage and travel little over the course of their life (Levin 1984; Levin et al. 1996), most estuarine-dependent species of nekton show large-scale habitat shifts through their ontogeny. In estuaries in the southeastern U.S. and Gulf of Mexico, many species migrate into estuaries as larvae

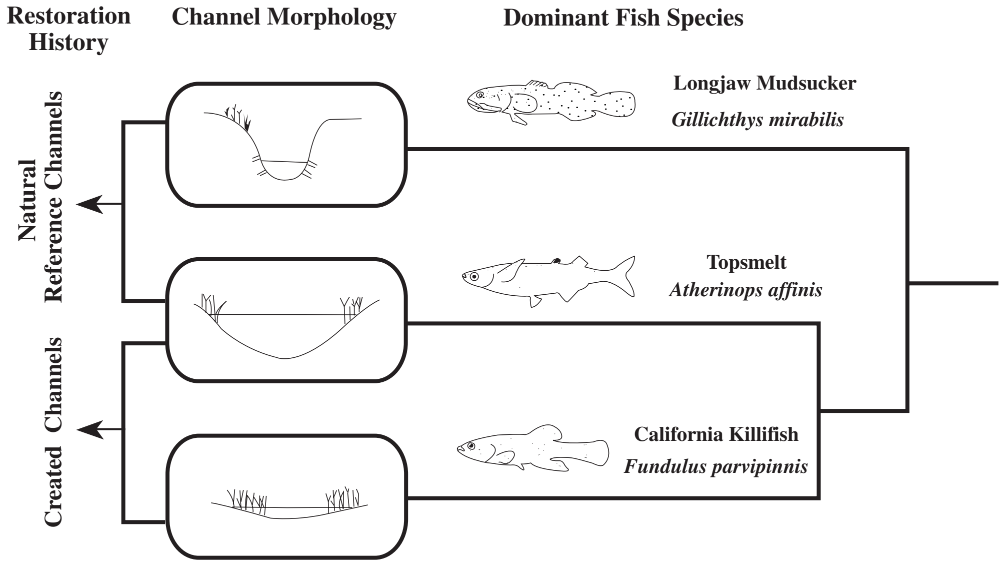

Figure 5.3 Relationship between channel restoration history, morphology, and dominant fish species in seine catches during summer sampling periods (1989–1996) in Sweetwater Marsh National Wildlife Refuge, San Diego, California, USA. (Modified from Williams and Zedler 1999, Fish assemblage composition in constructed and natural tidal marshes of San Diego Bay: Relative influence of channel morphology and restoration history. *Estuaries* **22**:702-716. With permission.)

or juveniles. Included are approximately two thirds of the commercially important species (Sciaenidae: red drum *Sciaenops ocellatus*, black drum *Pogonias cromas*, and spotted seatrout *Cynoscion nebulosus*; Bothidae: summer flounder *Paralichthys dentatus*). On the Pacific Coast, the English sole (*Pleuronectes vetulus*) and Dungeness crab (*Cancer magister*) both migrate from offshore habitats into estuaries as larvae or juveniles. Growth is accelerated during their first one or two years of life; they then return to the ocean (Gunderson et al. 1990). Diamond turbot (*Hypsopsetta guttulata*) and California halibut (*Paralichthys californicus*) use southern California bays as nursery areas, displaying distinct ontogenetic distributions and increasing length with depth (Kramer 1991).

Ontogenetic habitat shifts may also take place on a much smaller spatial scale within an estuary. Baltz et al. (1993) showed that the speckled worm eel (*Myrophis punctatus*) moved from shallow (mean depth 38 cm) habitats near the shore (mean distance 1.2 m) into deeper waters further from shore (mean depth, 44 cm; mean distance, 1.7 m) as they increased in size. Other species shifted into higher salinity habitats or areas with higher density vegetation. Kneib (1987) showed that in a Georgia salt marsh, young killifish (*Fundulus heteroclitus*) were predominantly located in shallow pools on the marsh surface, while adults were concentrated in deeper subtidal areas. Habitat selection was related to predation in both juveniles and adults. Young killifish avoided larger piscivorous fish by remaining on the marsh surface, while adults in subtidal areas could more effectively avoid avian predators.

Finally, animal movements over small temporal scales can link very different aspects of the diverse estuarine landscape. Many estuarine species depend on resources from both intertidal and subtidal habitats. These transient species use intertidal mudflats or the vegetated marsh surface at high tide to feed and/or avoid predators before retreating to subtidal channels at low tide (Miller and Dunn 1980, Wolff et al. 1981, Kneib 1987, Rozas and LaSalle 1990). Connectivity between, and proximity to, heterogeneous habitats influence species composition, fitness, and movement. For instance, Irlandi and Crawford (1997) found that abundance and growth of pinfish in (*Lagodon rhombiodes*) were higher in

intertidal salt marshes adjacent to seagrass beds than in unvegetated subtidal mudflats, and that the type of subtidal habitat present influenced pinfish movements. Location within an estuary and proximity to natural sources of immigrant populations also strongly influence colonization and community development (Bell et al. 1988, Sogard 1989).

Despite the importance of landscape factors to ecosystem function, restored habitats are often fragmented and spatially isolated (Simenstad and Thom 1996, Bell et al. 1997). Understanding the long-term viability of populations in these landscapes requires information about their relative isolation from other source populations, including colonization, migration, and extinction rates (Montalvo et al. 1997). If the marsh is far from a natural "source population," colonization may occur quite slowly. Organisms with a planktonic larval stage may recruit, but those with limited dispersal abilities may need to be seeded to promote their establishment (Levin et al. 1996). The infauna that colonized a planted *Spartina alterniflora* marsh in North Carolina 1 week to 27 months after construction had much greater dispersal ability than the infauna of a nearby natural marsh. Species lacking a planktonic larval stage (direct developers) were much slower to reach the planted marsh and made up only a small proportion of the abundance there (Figure 5.4; Levin et al. 1996). Restoring or constructing salt marshes in close proximity to natural marshes should accelerate the development of infaunal communities (Sacco et al. 1994). Direct water connections may be important for some taxa.

Marsh "edge" is the interface between vegetated marsh and open water; it provides a link between two habitats with distinct functions. Edges are gaining more recognition as studies document high densities of marsh macrofauna where vegetated intertidal habitats meet permanent subtidal habitats. In a study of microhabitat use by small fish in the Gulf of Mexico, Baltz et al. (1993) found that the 15 most abundant fishes were concentrated within 1.25 m of the marsh edge. Peterson and Turner (1994) also found that densities of most organisms (both fish and decapod crustaceans) in a Louisiana salt marsh were highest within 3 m of the marsh edge. Edge habitat appeared to be particularly important to transient, estuarine-dependent species, such as spotted seatrout (*Cynoscion nebulosus*) and penaeid shrimps (*Penaeus aztecus* and *P. setiferus*). In created *Spartina alterniflora* salt marshes of Galveston Bay, Minello et al. (1994) observed that shrimp densities were 5 to 13 times higher near the edges of experimental channels. The greatest length of marsh edge will occur when topography is heterogeneous; hence channels and creeks should meander, rather than have straighter canal-type configurations.

All of these results suggest that habitat values can be enhanced by incorporating complex creek networks into construction designs for tidal marshes. In southern California, constructed systems are usually much simpler than natural systems, with one or two large subtidal creeks or, in some cases, subtidal basins (Figure 5.5). Because the majority of marsh-water interface in California marshes is contributed by the smallest tidal creeks (Pestrong 1965, Coats et al. 1995, Desmond et al. 2000), constructed marshes lacking these creeks may not provide the same habitat functions as natural marshes. Landscape-based design decisions can have notable implications for restoration and mitigation projects. Furthermore, the proximity of a restoration site to other estuarine habitats can significantly influence faunal colonization and assemblage development.

5.3 *Estuarine habitat assemblages, functions, and restoration*

Physical forces create marsh habitats with morphological variability, accentuated by differences in sediment composition, vegetation, and climate (Eisma et al. 1997). While marsh habitats form a continuum, their use by fishes and invertebrates and the measuring of associated attributes are most easily understood by imposing some level of habitat categorization (e.g., Able et al. 1996). This section is subdivided into five habitat categories

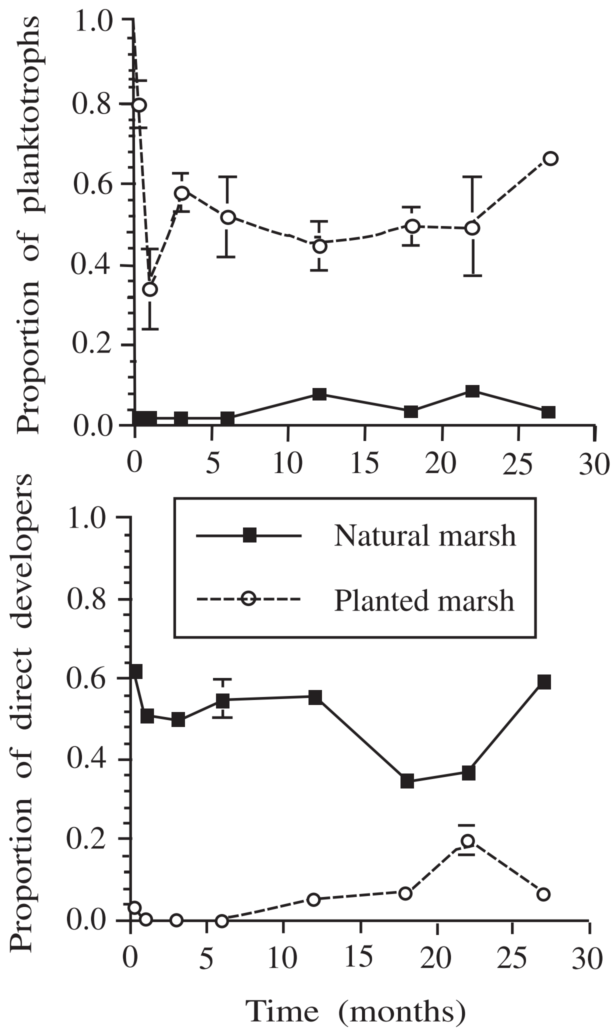

Figure 5.4 Proportional representation of individuals with different developmental modes in natural and planted *Spartina alterniflora* marshes in the Newport River Estuary, North Carolina, USA. Direct developers are considered dispersal-limited and were abundant only in the natural marsh; planktrotrophs (which have a feeding planktonic larval stage) were very common in the planted marsh. (Modified from Levin et al. 1996, Succession of macrobenthos in a created salt marsh. *Marine Ecology Progress Series* **141**:67-82. With permission.)

with descriptions of typical fish and invertebrate assemblages, functions provided by the habitat, and examples of restoration efforts to date. Descriptions of each habitat type can be found in Chapter 2 (Box 2.5).

5.3.1 Subtidal assemblages

Subtidal areas have a high degree of habitat complexity over many scales and generally support a diverse fish and invertebrate assemblage composed of resident species as well as seasonal marine migrants. In the interest of brevity and relevance, we limit our discussion to the relationship between subtidal and marsh habitats; specifically, the functional

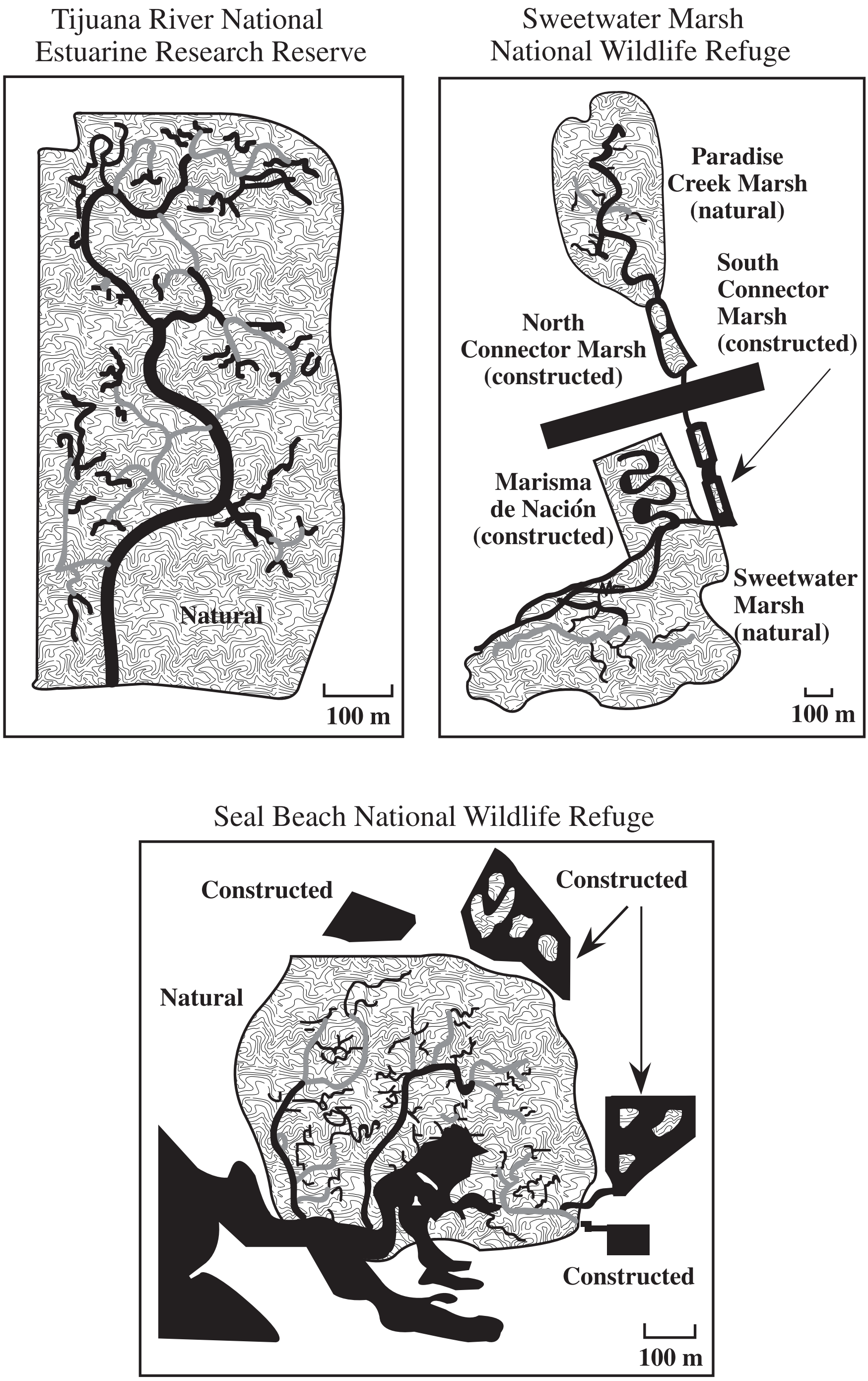

Figure 5.5 Geomorphic complexity of natural and constructed wetlands in southern California, USA. Creek networks in natural wetlands (Tijuana Estuary and Seal Beach) are color-coded by order: third and fourth-order creeks are bold black, second-order creeks are bold gray, and first-order creeks are plain black. Marisma de Nación and North and South Connector Marsh were constructed in 1985 and 1990, respectively, within the Sweetwater Marsh National Wildlife Refuge in San Diego Bay; the constructed wetlands around the periphery of the natural marsh at Seal Beach National Wildlife Refuge in Seal Beach were completed in 1990.

interactions between these habitats. Seagrass beds, for example, share many species with higher elevation marsh habitats and are recognized as key areas that provide subtidal refugia and foraging.

Typically, more species and families of nektonic fishes and invertebrates occur where subtidal habitats (open impoundments, embayments, canals, etc.) have a continuous connection to tidal marsh habitats at all tidal stages (Kneib 1997). The high species richness in these habitats is due primarily to the number of marine transients. However, the most abundant fish species in many subtidal estuarine habitats are seasonal residents (e.g., schooling baitfish such as atherinids, engraulids, and clupeids), which migrate offshore during the winter, and resident species (e.g., cyprinodontids), which spend their entire life in the estuary (Horn and Allen 1985, Able et al. 1996, Kneib 1997). Invertebrates populating these habitats may include marine species, such as polychaetes, sea stars, sand dollars, nudibranchs, crustaceans (especially penaeid and palaemonid shrimps, portunid crabs), and a variety of bivalves and gastropods (e.g., see Ricketts and Calvin 1968, Kneib 1997).

5.3.2 Subtidal function

Subtidal habitats, including bays and harbors, moderate the influence of the ocean and provide a subtidal refuge for a number of species. Accordingly, these habitats are generally considered important nurseries, because their high productivity and structural complexity provide both enhanced foraging opportunities and predator refugia for juvenile marine fishes and crustaceans, thus leading to rapid growth and high production (e.g., Turner 1977, Weinstein 1979, Allen 1982, McIvor and Odum 1988).

Subtidal habitats (especially shallow vegetated margins) may also serve as intermediate transition zones, functioning as staging areas and migration routes for various life history stages of species moving between marine, estuarine, and/or riverine habitats (e.g., salmonids, Simenstad et al. 1982). Fish movement between marsh edge and subtidal seagrass beds can provide an important trophic linkage between habitats, allowing the transfer of marsh- or seagrass-derived production between these areas and enhancing growth of individuals employing this strategy (Weinstein and Brooks 1983, Irlandi and Crawford 1997). While the vegetated shallows may provide refuge from larger piscivorous fishes, deeper subtidal habitats reduce predation by birds (e.g., egrets and herons) (Kneib 1987). For many estuarine residents (e.g., *Hypsopsetta guttulata*, *Paralabrax* spp.), deeper subtidal regions and bay mouths are the primary habitat for adults, and they function as spawning and reproductive habitats (Gunderson et al. 1990, Emmett et al. 1991, Kramer 1991, Allen et al. 1995). In southern California, subtidal bays and lagoons also appear to serve as warm-water refuges for some subtropical marine species, such as the recreationally important spotted sandbass (*Paralabrax maculatofasciatus*) (Allen et al. 1995).

5.3.3 Subtidal restoration

In southern California, subtidal basins are included where the restoration goal is to replace lost deepwater habitat (e.g., from the construction of marina or port facilities). Although deepwater basins are readily colonized by fish, they are often excavated from upland or from filled wetlands; thus, they may lack connections to intertidal habitats. In 1994 a rectangular subtidal basin was excavated from a large fill area within the Sweetwater Marsh National Wildlife Refuge to compensate for damages to fish habitat elsewhere in San Diego Bay. The 1-ha project was surrounded by upland on three sides with extremely steep slopes and little intertidal transition habitat (Figure 5.6). This mitigation project was not designed to be an integral part of the landscape and its animal assemblages were not assessed; as a result, its subtidal habitat values remain questionable. Similarly, in 1990 four

Figure 5.6 D Street fill restoration project, a "basin-type" excavation with little intertidal linkage, which was intended to provide habitat for estuarine fishes. San Diego Bay, California, USA.

basins totaling 46.2 ha were excavated within the Seal Beach National Wildlife Refuge in Anaheim Bay to compensate for the filling of a nearby deepwater harbor area. The four basins were excavated from upland and historical wetland areas and were connected to tidal areas by a system of culverts (MEC Analytical Systems, Inc. 1995, Zedler et al. 1997). This project, like the 1-ha basin, was not natural in its shape or location, and it had limited connectivity with shallow intertidal habitats. While projects of this type provide habitat for some species or adult life history stages, they may lack essential functions for species and life history stages that depend on shallow marsh habitats (Ruiz et al. 1993, Baltz et al. 1993).

5.3.4 Intertidal flat assemblages

The most visible, surface-dwelling fauna of intertidal flats are gastropod molluscs and crabs. In southern California, common species are the crabs *Uca crenulata*, *Pachygrapsus crassipes*, and *Hemigrapsus oregonensis*, and the horn snail, *Cerithidea californica*, which can reach densities of 1000 per square meter (Warme 1971; Figure 5.7). The less visible benthic infauna include oligochaetes, polychaetes such as *Capitella* spp. and *Streblospio benedicti*, amphipods, filter-feeding bivalve mollusks, and thallasinidean shrimp. Although most fish can only use these habitats when they are flooded, burrowing gobiids, such as *Clevelandia ios*, can reside in the intertidal mudflats throughout the tidal cycle (Brothers 1975, MacDonald 1975).

5.3.5 Intertidal flat function

Deposit-feeding invertebrates find an abundant food source in sediments and in micro- and macroalgae; the latter are often abundant on the mudflats because of high light availability (Zedler 1980). Dense invertebrate assemblages, in turn, provide food for shorebirds, fish, and crabs (Peterson 1981, Quammen 1984). Intertidal sand and mudflats are primary feeding areas for both resident and migratory shorebirds and waterfowl in many coastal habitats (Recher 1966, Boland 1981, Way 1991, Burger et al. 1997; Figure 5.8). Filter-feeding invertebrates are also well served by the high water flow rates of mudflats; the extensive mudflats of the Pacific northwest are valued for oyster growing (Simenstad and Fresh 1995). In addition, many fish, including commercially important species such as winter flounder (Wells et al. 1973), plaice (Kuipers 1973), and English sole (Toole 1980), migrate from subtidal

Figure 5.7 Intertidal mudflats, Tijuana Estuary, California, USA. The abundant snail is *Cerithidea californica*; the crab is *Uca crenulata*.

Figure 5.8 Shorebird use of mudflat habitats at Tijuana Estuary.

areas to intertidal mudflats at high tide. Migrations are primarily related to feeding (Miller and Dunn 1980, Wolff et al. 1981, Van der Veer and Witte 1993), but seasonal movements may also serve a predator-avoidance function (Kneib 1987, Ruiz et al. 1993).

5.3.6 Intertidal flat restoration

There are few published studies on constructed mudflats. Lee et al. (1998) showed that constructed intertidal flats in Japan were characterized by many of the problems that affect constructed salt marshes; constructed intertidal flats had lower contents of silts, nitrogen, and organic matter than natural intertidal flats.

The loss of tidal flats in southern California has been even greater than that of tidal salt marshes (Macdonald 1990), primarily because of human activities such as dredging and filling. However, unvegetated intertidal habitats are often omitted in tidal wetland restoration plans. More attention is given to vegetated habitats, which are valued for their support of endangered birds (e.g., *Spartina foliosa* for light-footed clapper rail nesting and *Salicornia virginica* for Belding's Savannah sparrow nesting) (Zedler 1996*a*). In estuaries of the Pacific Northwest and in Great Britain, a major concern is loss of mudflat area following invasion by introduced or hybridized species of *Spartina* (Harrington and Harrington 1992). Because of extensive losses, mudflat restoration should be given a higher priority in future projects.

5.3.7 Channel assemblages

Channels, which we define as having order 3 or higher (Strahler 1964; see Chapter 2, Box 2.5), have a higher edge:volume ratio than subtidal basins, and they support a greater proportion of shallow water species. Transient marine species that are not usually found in intertidal areas may be found in subtidal channels. Able et al. (1996) showed that the diversity of subtidal channel fish fauna was enhanced both by a number of shallow-water families (Atherinidae and Cyprinodontidae) and marine migrants (e.g., Clupeidae, Engraulidae, Syngnathidae, Carangidae, Sciaenidae). At Tijuana Estuary, California halibut (*Paralichthys californicus*) were consistently more dense in broad, deep channels close to the estuary mouth, which had sandy sediments, gradually sloping banks, and high tidal flows (Williams et al. 1998*b*). This marine species uses nearshore and estuarine habitats as nursery areas (Kramer 1991). The channel invertebrate assemblage is similar to that of the mudflat, with snails, crabs, bivalves, and burrowing shrimp dominating the biomass and polychaetes and amphipods dominating numerically.

5.3.8 Channel function

Tidal channels connect salt marshes to subtidal basins and to the ocean, serving as a conduit for animals, energy, and material. Channels provide access to the marsh surface, pathways for migration, links for species using multiple habitats, and transition areas for larvae and juveniles (Nordby 1982). Marshes with abundant channels have high edge:volume ratios, providing more shelter and refuge for fishes and invertebrates (Minello et al. 1994). Because channels retain a substantial portion of their volume at low tide (Figure 5.9), they also serve as a low-tide refuge for estuarine organisms, and their banks provide areas for shorebird foraging (Boland 1981).

5.3.9 Channel restoration

At Sweetwater Marsh in San Diego Bay, constructed marsh channels were monitored and compared with nearby natural habitats three years after excavation. Although the sediments of the constructed marsh were much sandier than those of the natural marsh (Langis et al. 1991), the channel invertebrates met the mitigation requirements for density and species richness (75% of the mean density and species richness of benthic invertebrates

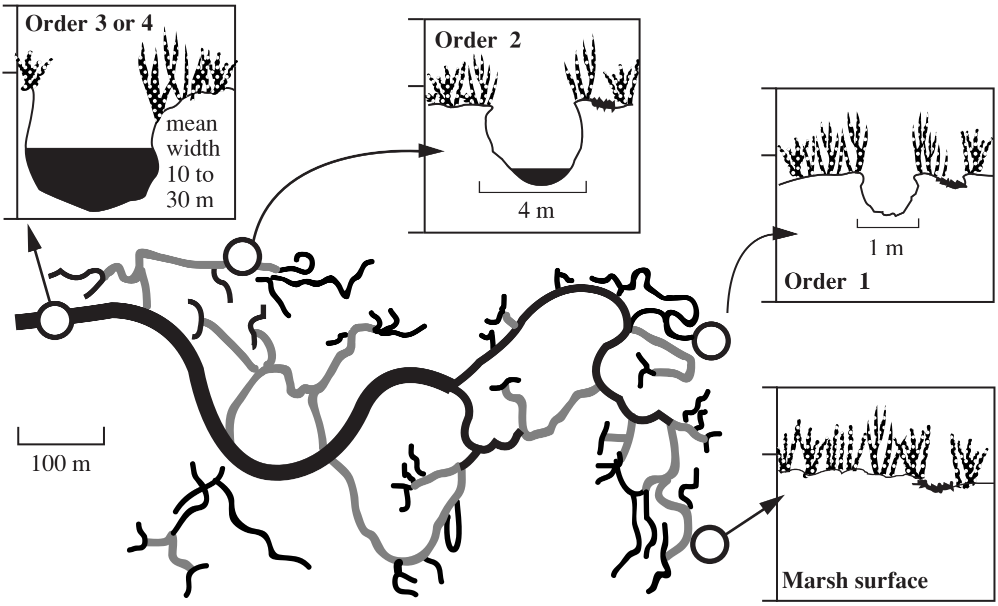

Figure 5.9 Creek network of Tijuana Estuary. Creeks are color-coded: third- and fourth-order creeks are in bold black, second-order creeks are in bold gray, and first-order creeks are in plain black. Insets show the typical profiles and widths of these creeks, and their condition at low tide. Higher creek orders (third- or fourth-order) retain a substantial portion of their volume at low tide; second-order creeks retain shallow pools at low tide, and first-order creeks and the marsh surface drain completely at low tide.

in natural channels) by the third year of monitoring (six years after excavation, PERL 1990). At two sites in North Carolina, there was greater faunal similarity between constructed and natural creek stations than between constructed and natural marsh surface stations (Cammen 1976). It is possible that at constructed marshes, invertebrate communities develop more rapidly in channels than on the marsh surface (Section 5.3.13). Dispersal and settling may occur more readily in channels, which receive more exposure to tidal action (and carry more planktonic larvae) than the marsh surface.

Fish assemblages in channels constructed at Sweetwater Marsh also quickly met mitigation requirements (Williams and Zedler 1999). Species richness and density in the constructed channels reached 75% of the natural channels within two years of construction. However, further analysis of the assemblages at natural and constructed sites showed that channel morphology had a strong influence on species composition (Section 5.2.5; Figure 5.3).

Several studies in the Pacific Northwest have examined the functions that constructed estuarine channels provide for juvenile salmon. These studies have indicated that juvenile chum (*Oncorhynchus keta*) and chinook (*O. tshawytscha*) salmon temporarily resided in the 1 to 2-year old restored channels during their outmigration and used them to forage on a variety of insects (Shreffler et al. 1990, 1992). Residence times and growth did not vary between natural and restored channels (Miller and Simenstad 1997) although diets were slightly different, possibly due to varying prey availability and/or foraging efficiency.

In general, it appears that invertebrate and fish communites in constructed channels can resemble those of natural sites, if conditions such as hydrology and channel morphology are appropriate. However, channels excavated at the wrong elevation/depth or with poor tidal exchange may develop algal blooms, anoxia, or sedimentation problems.

5.3.10 Creek assemblages

When flooded, intertidal marsh creeks (order 1 or 2; Strahler 1964) are used extensively by fish in marshes along the U.S. Atlantic (Cain and Dean 1976, Shenker and Dean 1979, Weinstein 1979, Rozas and Odum 1987*b*, Sogard and Able 1991) and Gulf coasts (Rozas 1992, Minello et al. 1994), as well as in Europe (Kelley and Reay 1988, Cattrijsse et al. 1994, Feunteun and Marion 1994) and South Africa (Paterson and Whitfield 1996). Fish species composition in creeks is similar to that on the marsh surface. Southern California intertidal (first-order) creeks are dominated by resident gobies (Gobiidae) and killifishes (Cyprinodontidae) (Desmond 1996); second-order creek samples contain these species as well but are dominated by a transient silverside (Atherinidae), *Atherinops affinis*. Creeks in New Jersey coastal marshes are dominated by a variety of smaller species (especially Atherinidae and Cyprinodontidae) and juveniles of larger species (e.g., Mugilidae, Able et al. 1996). Little information is available on the invertebrate assemblages of small tidal creeks.

5.3.11 Creek function

Although first- and second-order creeks in many geographic regions are almost completely drained at low tide (Figure 5.9), they expand the area available for foraging by estuarine fish at high tide when they are flooded (Fitz and Wiegert 1991, Rountree and Able 1992, Allen et al. 1994). They may also serve as nursery habitats, as the majority of fishes collected there are juveniles (Shenker and Dean 1979, Weinstein 1979, Weinstein and Brooks 1983, Desmond et al. 2000). In North America's Atlantic and Gulf Coast wetlands, the first-order creeks appear to be preferred habitats for some resident species; several studies show that densities of fish increase as stream order decreases (Weinstein 1979, Hettler 1989, Rozas and Odum 1987*a*, Rozas 1992). More recently, Rozas et al. (1988) found that first-order creeks serve as conduits for fishes moving onto the marsh surface at high tide; fishes are more likely to travel across first-order creeks than across higher-order channel banks. Thus, shallow creeks promote tidal access for fishes to low elevation intertidal areas, where primary production is high (Zedler 1980). Because of their slow flows, small intertidal creeks may accumulate heavy macroalgal biomass (Rudnicki 1986); this may provide reproductive areas for species such as topsmelt (*Atherinops affinis*), which attach their eggs to algae.

5.3.12 Creek restoration

Small creeks are rarely included in salt marsh restoration designs, and their absence may reduce the fish nursery function of restored marshes. Predation rates for juveniles of most estuarine species are known to decrease with decreased water depth (McIvor and Odum 1988, Ruiz et al. 1993), and marshes without shallow creeks have little shallow water habitat. In a constructed marsh in Virginia, the lower density of blue crabs (relative to reference sites) was hypothesized to be caused by decreased habitat complexity in the new marsh, which lacked small first- and second-order creeks (Havens et al. 1995).

5.3.13 Marsh surface assemblages

Animal assemblages of the marsh surface are composed of both resident and transient species. The resident marsh epifaunal assemblages are dominated by gastropod and bivalve molluscs (e.g., mytilids), insects, amphipods, isopods, and crabs (Subrahmanyam and Coultas 1980, Scatolini and Zedler 1996). Dense populations of oligochaetes (LaSalle et al. 1991, Sacco et al. 1994, Levin et al. 1998) and polychaete worms (Lana and Guiss 1992, Minello and Zimmerman 1992) are also found within the sediments. Transient nekton generally access the marsh surface when it is flooded during high tide events, but large

numbers may also remain in shallow pool microhabitats when they are available (McIvor and Odum 1988, Hettler 1989, Kneib 1997). The few fish families that dominate these habitats include killifishes (Cyprinodontidae), silversides (Atherinidae), and gobies (Gobiidae) in mid-Atlantic coast marshes (Able et al. 1996) and killifishes, gobies, livebearers (Poeciliadae), and drums (Sciaenidae), along with the nektonic crustacean families Palaemonidae, Penaeidae, and Portunidae in south Atlantic coast marshes (McIvor and Rozas 1996, Kneib 1997). The few studies of fish use of Pacific coast marshes (Chamberlain and Barnhart 1993, Williams et al. 1998*a*, West and Zedler *in press*) suggest similar patterns by many of the same fish families (Cyprinodontidae, Atherinidae, Gobiidae).

5.3.14 Marsh surface function

The functional benefits of the vegetated intertidal marsh to animal assemblages has been fairly well documented, especially along the southeast and Gulf coasts of North America (Kneib 1997). Nektonic fish and crustaceans migrate onto the marsh with the tides to feed, and commonly have greater gut fullness at high or ebbing tides (Harrington and Harrington 1961, McIvor and Odum 1988, Rozas and LaSalle 1990, Rountree and Able 1992, Kneib 1997). At San Diego Bay, California killifish (*Fundulus parvipinnis*) allowed to move onto a *Salicornia virginica*-dominated marsh at high tide had fuller guts (Figure 5.10) and more varied diets than fish prevented from accessing the marsh (West and Zedler *in press*; also see Box 5.1). Fish with access to the marsh had more isopods, insects, and gastropods (*Assiminea californica*) than fish limited to the channels (Figure 5.11); fish in the channels fed predominantly on amphipods, polychaetes, ostracods, and detritus. Food items include not only resident marsh biota, but also detritus derived from decomposition of marsh vascular plant production. Even

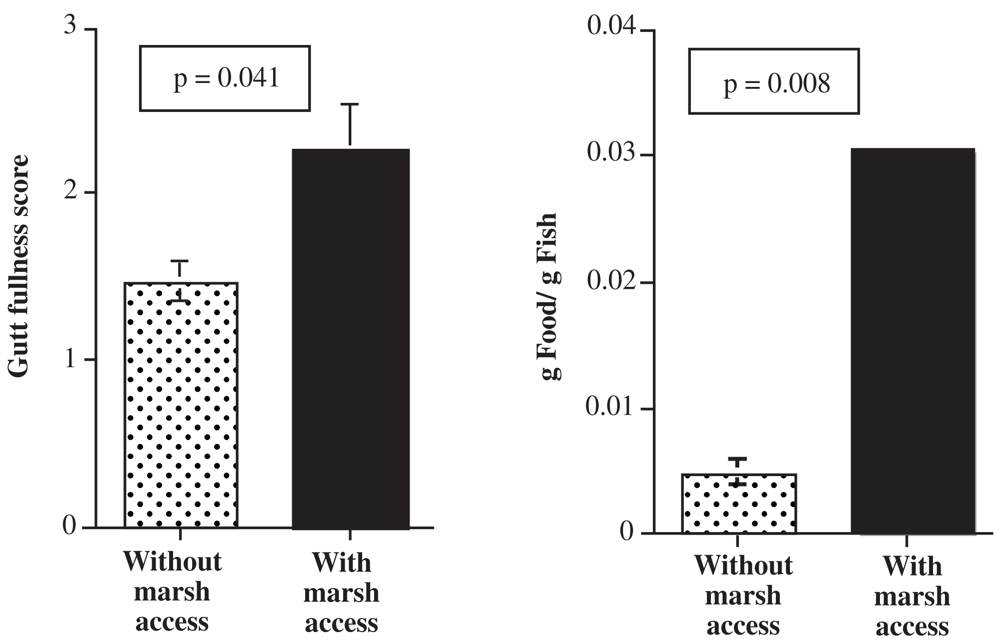

Figure 5.10 Summary of gut contents of California killifish (*Fundulus parvipinnis*) with and without access to pickleweed (*Salicornia virginica*) marsh in San Diego Bay, California, USA. Gut fullness represents a qualitative score assigned at the time of dissection, where 0 = empty, 1 = 25% full, 2 = 50% full, 3 = 75% full, and 4 = completely full. Weight-specific consumption (wet weight of gut contents/wet weight of the fish) is a quantitative measure of gut fullness. Error bars represent ±1 standard error. Results of Student's t-tests are shown as p values (J. West, unpub. data).

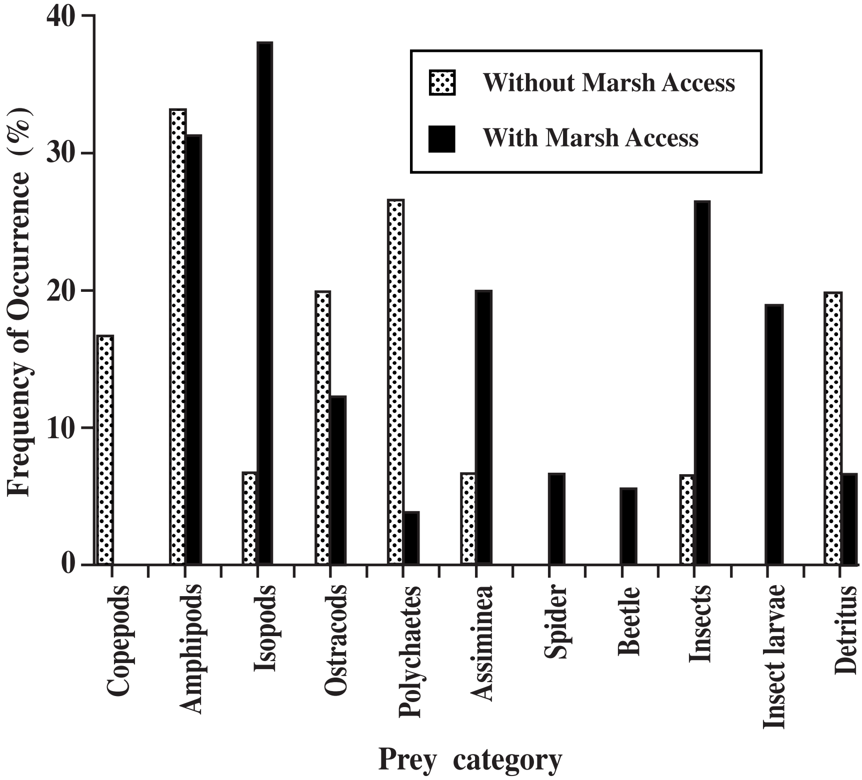

Figure 5. 11 Frequency of occurrence of prey items found within guts of California killifish (*Fundulus parvipinnis*) with and without access to pickleweed (*Salicornia virginica*) marsh in San Diego Bay, California, USA. Fish were collected between June and September 1997 (Modified from Johnson 1999, *Fish use of a Southern California salt marsh.* Master's thesis. San Diego State University, San Diego, California, USA).

species that do not use marsh areas directly may benefit from their presence. Marsh primary producers (both vascular plants and benthic microalgae) are known to fuel estuarine food webs on the Atlantic (e.g., Teal 1962, Peterson and Howarth 1987), Gulf of Mexico (Sullivan and Moncreiff 1990), and Pacific Coast (Kwak and Zedler 1997).

Marsh vegetation provides both resident and transient species with a partial refuge from larger fish predators (Ryer 1988). This may explain the occurrence of high densities of small crustaceans, such as juvenile penaeid shrimp and portunid crabs, in vegetated marsh habitats (Zimmerman and Minello 1984, Kneib 1984). Likewise, animals able to use higher marsh elevations gain a refuge from transient predators faced with tidally induced time constraints on foraging. The abundance and size distribution of some molluscs and crabs has been related to an intertidal gradient in predation intensity (Hamilton 1976, Wilson 1989, Peterson and Turner 1994).

A number of transient (mostly atherinids) and resident (cyprinodontids) species use the intertidal marsh surface to spawn (Kneib 1997). Most of these species have a spawning periodicity that coincides with the lunar cycle and tidal events. Eggs, which are often dessication-resistant, are often deposited on high spring tides in protected areas (e.g., algae mats, *Spartina* stems, shells, sediment), where they develop for 7 to 14 days before hatching on the subsequent spring tide series. Juvenile forms of some species (mostly penaeid and palaemonid shrimp, and cyprinodontids) use shallow pools of the marsh surface as a nursery area (Kneib 1984, Talbot and Able 1984).

Box 5.1 The value of vegetated marshes to fish: predictions of growth using a bioenergetics model

Sharook P. Madon

Fish growth is a desirable indicator of aquatic habitat function because it integrates the effects of all abiotic and biotic factors. It is difficult to measure fish growth directly, but it can be estimated as the difference between energy consumed in food and energy lost via egestion, excretion, respiration, and specific dynamic action (SDA, the costs of processing food). Bioenergetics models summarize the fish growth process in a balanced energy budget, where growth equals that difference (Kitchell 1983, Hewett and Johnson 1987, Rice 1990, Hanson et al. 1997). Specific consumption and respiration rates (with units of Joules/gram of fish/day) are modeled as non-linear responses to fish body mass and water temperature. Egestion, excretion, and SDA, however, are often set as constant proportions of energy consumed (Hanson et al. 1997). As such, bioenergetics models explicitly link fish energetic processes to biotic (food resource, foraging constraints, physiological condition) and abiotic variables (water temperature, salinity, pollutants). Thus, they are versatile tools for evaluating processes that affect fish growth.

The California killifish, *Fundulus parvipinnis*, uses vegetated marsh surfaces for feeding. Killifish that feed on the marsh surface have fuller guts (on average six times fuller on a g/g basis) than those feeding exclusively in subtidal channels, and they take advantage of foods such as insect larvae that are not available in channels (West and Zedler *in press*). However, access to salt marshes will translate into higher growth rates only if (1) the marshes are available to fish for sufficiently long time periods and (2) if the energy cost of swimming onto the marsh is low relative to that of the food consumed. The hydroperiods of southern California tidal marshes contrast with those on the Gulf and Atlantic coasts; the region's semidiurnal tides allow relatively limited marsh access throughout the year. With short periods of access, the energetic costs of swimming up to, and foraging on, the marsh surface may be high, in part because the water becomes warmer and respiration rates increase. Also, the assimilation of diets of insect larvae and other marsh-derived foods may be low, negating some or all of the potential growth advantage. Bioenergetics models can evaluate the effects of a six-fold increase in energy intake, the increased energy costs of swimming in warmer water and the assimilation of less of the food, thus simulating killifish growth during periods of marsh access.

We use a *Fundulus* bioenergetics model (Hanson et al. 1997; Table 5.1) to explore whether or not limited marsh access enhances the growth of California killifish. We start with an individual killifish of 30 mm TL (total length), with a wet weight of approximately 0.45 g. Growth is then calculated over 1 yr under simulated conditions with and without marsh access and at water temperatures and hydroperiods typical of southern California (West and Zedler *in press*). Our field data are used to set killifish daily food consumption rates from 3 to 9% of body mass (g/g/d), depending on fish size and time of year. During each 2- to 7-day period each month that killifish have marsh access, we double its daily food intake rate, increase its metabolic costs by 25%, decrease its food assimilation by 10%, and increase water temperatures by 2°C.

Despite limited access to the marsh, lower food assimilation, and higher metabolic costs, the bioenergetic model simulates 30% more growth for killifish with marsh access than for those completely restricted to subtidal channels. Because killifish use the tidal flow in rivulets to carry them onto the marsh (Desmond et al. 2000), and because insects

Table 5.1 Parameters for the mummichog, *Fundulus heteroclitus*, bioenergetics model that was used to simulate growth of the California killifish, *Fundulus parvipinnis*. Parameters represent all life stages and were estimated from Targett (1978), Weisberg et al. (1981), Weisberg and Lotrich (1982), and Kneib and Parker (1991).

Symbol	Parameter Description	Parameter Value
	Consumption	
a_c	Intercept for maximum consumption (g g^{-1} d^{-1}, wet wt.)	0.20
b_c	Exponent for maximum consumption	–0.25
Q_c	Slope for temperature-dependence of consumption	2.22
T_{copt}	Optimum temperature for consumption for larvae	30°C
	for adults	27°C
T_{cmax}	Maximum temperature for consumption	34°C
	Respiration	
a_T	Intercept for routine respiration (g O_2 g^{-1} d^{-1})	0.0203
b_r	Exponent for routine respiration	–0.174
Q_r	Slope for temperature-dependence for routine respiration	2.0
T_{ropt}	Optimum temperature for routine respiration	29°C
T_{rmax}	Maximum temperature for routine respiration	36°C
S	Specific dynamic action coefficient	0.10
A	Activity parameter	1.25
	Egestion and excretion	
a_f	Proportion of consumed food egested	0.10
a_u	Proportion of assimilated food excreted	0.06
	Caloric densities (calories g^{-1} wet wt.)	
	Larvae	1000
	Juveniles/adults	1200
	Zooplankton prey	774
	Benthic prey	1000

form a relatively small proportion of their total diet (West and Zedler *in press*), we consider our growth estimates to be conservative; that is, we overestimate energy costs and underestimate food assimilation.

These results support our hypothesis that killifish benefit from sporadic foraging on marshes. Future experiments in aquaria will test further the controls on fish foraging and growth. Collectively, our findings will provide key scientific input for two policy questions: "How much mitigation credit should be given for including salt marshes in fish-habitat restorations?" and "How much mitigation credit should be given for in-kind vs. out-of-kind mitigation?"

5.3.15 Marsh surface restoration

Several studies have focused on invertebrates of constructed marsh habitats, particularly resident infaunal communities. *Spartina alterniflora* marshes near Chesapeake Bay in Virginia (Havens et al. 1995) and Winyah Bay, South Carolina (LaSalle et al. 1991), had infaunal species richness and density similar to nearby natural marshes within 5 years of construction. In contrast, constructed *S. alterniflora* marshes in Galveston Bay, Texas, had lower infaunal density and species richness than natural reference marshes up to 15 years

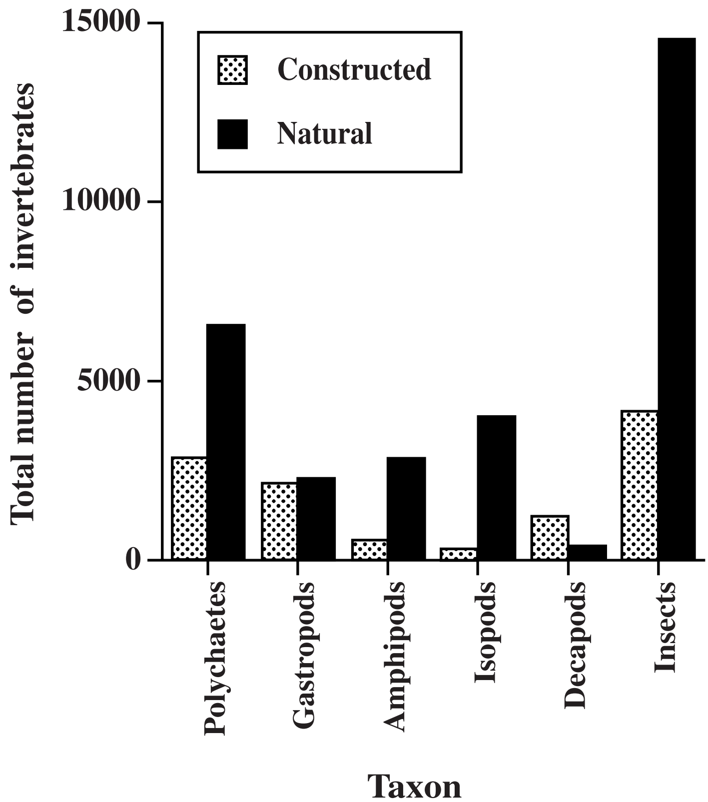

Figure 5.12 Abundance of major groups of epibenthic invertebrates collected from litterbag traps in natural and 4-year-old constructed salt marshes over eight sample dates between May 1988 and March 1989 in San Diego Bay, California, USA. (Modified from Scatolini and Zedler 1996, Epibenthic invertebrates of natural and constructed marshes of San Diego Bay. *Wetlands* **16**:24-37. With permission.)

after construction (Minello and Webb 1997). Six constructed *S. alterniflora* marshes along the North Carolina coast (1 to 17 years old) also had lower densities than natural reference marshes, although species richness and composition were similar (Sacco et al. 1994). At another constructed *S. alterniflora* site in North Carolina (Newport River Estuary), species richness and density were similar to nearby natural marshes within 6 months of excavation, but differences in species composition and trophic groupings remained 4 years afterwards (Levin et al. 1996). The variation between sites of different ages and in different locations has mostly been related to the environmental parameters described above (Section 5.2), including hydrology, elevation, and substrate characteristics.

Epifaunal invertebrate communities may also differ between constructed and natural wetlands. Four-year old constructed *Spartina foliosa* marshes in San Diego Bay had approximately one third the population of epifauna of nearby natural marshes, with amphipods, isopods, and insects occurring in particularly low numbers (Scatolini and Zedler 1996) (Figure 5.12). Lower densities were attributed to lower sediment organic matter resources, as well as to potential disturbance effects of burrowing shorecrabs, *Hemigrapsus oregonensis*, which colonized the restored sites early on and were much more abundant there than at natural sites. Amphipods, isopods, and insects are prominent in the diets of fish which migrate onto nearby marshes to feed at high tide (West and Zedler *in press*), suggesting that reduced diversity or density of epifauna at constructed sites may decrease fish food web support functions.

Studies of fish use of vegetated marshes have generally shown lower abundance in constructed than natural sites (Moy and Levin 1991, Chamberlain and Barnhart 1993, Minello and Webb 1997). Differences in abundance are usually attributed to differences in marsh elevation, flooding, and *Spartina* stem densities.

5.4 Summary

This chapter reviews the patterns and potential mechanisms of macroinvertebrate and fish species distribution and abundance in natural and created marshes, with an emphasis on examples from southern California. In these aquatic systems, which have frequent tidal influence, hydrology and topography are the primary forces that structure and characterize habitat features, which in turn affect patterns of use by associated biota. Consequently, it is critical to devote attention to these factors in the earliest stages of restoration planning in order to provide for the natural development and sustainability of the associated biotic community.

Restored wetlands offer many opportunities to explore the functional importance of microhabitats, as manipulative experiments can be built into the site design, and large-scale investigations can be accomplished with replication (e.g., Langis et al. 1991, Zedler et al. 1992, Gibson et al. 1994, Zedler 1996*b*, Mitsch and Wilson 1996). Restored marshes should play a larger role in helping scientists understand the relationships between marsh characteristics and invertebrate and fish abundance, species composition, and colonization (see Box 5.2) (Palmer at al. 1997). Likewise, they can be useful tools to further explore challenging ecological questions of marsh production and trophic support pathways, landscape connectivity and complexity, and exotic species invasion and establishment (Zedler et al. 1997).

A number of questions related to the ecology and restoration of marshes and associated fauna remain. For example, how do landscape scale, habitat fragmentation, and habitat interactions influence animal movements and community sustainability? How do habitat linkages influence rates of animal colonization at restored sites? How do invasive, exotic organisms affect native species? Invasive species are a major issue for the San Francisco Bay (Nichols and Thompson 1985, Meng et al. 1994) and for the Bay Delta, where major efforts are being planned to restore sensitive habitats. Finally, how do anthropogenic disturbances and contamination affect the establishment and persistence of wetland biota? Urban environments present a major challenge because they lack pristine reference sites, impairing our ability to establish restoration benchmarks (Zedler 1996*a*). We believe that adaptive restoration and detailed, long-term monitoring of restored sites will help answer many of these questions.

Box 5.2 Fish and invertebrate colonization rates in restored/created salt marsh channels

Gregory D. Williams and Julie S. Desmond

Hypothetical trajectories for restoration sites suggest smooth, rapid development, eventually matching natural reference sites over time (see review in Zedler and Callaway 1999). Several studies of created marshes also indicate that fish and benthic invertebrate assemblages

develop quickly through natural recolonization. For example, several researchers have observed that the density of some benthic macrofauna progressively increases with constructed marsh age (Lasalle et al. 1991, Minello and Zimmerman 1992, Posey et al. 1997). The density and standing stock of fish that use newly constructed channels may also increase over time (Simenstad and Thom 1996).

Our detailed monitoring of restoration sites at Marisma de Nación (MN) in San Diego Bay's Sweetwater Marsh National Wildlife Refuge (Box 1.8) and the Tidal Linkage (TL) in the north arm of Tijuana Estuary (Box 1.10) provide a different perspective. At both sites, the response of invertebrate and fish assemblages (that is, change in species richness and density) was not linear, nor was it necessarily additive. Both sites (especially TL) experienced obvious changes through time in channel morphology due to sedimentation and hydrologic processes. The observed rate of faunal assemblage development was highly related to these changing site characteristics, and the rate varied by taxonomic group, assessment parameter, and monitoring time period.

We found that small, tube-dwelling invertebrates (spionid polychaetes and corophiid amphipods) colonized new sites within a few weeks following excavation, initially dominating the benthic invertebrate assemblage (Figure 5.13). At MN, corophiid amphipods and tanaids exhibited brief population peaks within 6 months of excavation. Although they declined over the next 2 years, we observed unusually large peaks 2 to 3 years later. Longer-lived, deeper-dwelling molluscs (e.g., California jack-knife clam, *Tagelus californianus*) did not colonize MN until 1991 (1.5 years after excavation); after this time, molluscs gradually increased in density. At this site, total density was generally higher than the mean density of Sweetwater Marsh reference sites. In contrast, invertebrate density was generally lower at TL than at Tijuana Estuary reference sites. Although the same "colonizer" taxa (spionid polychaetes, corophiid amphipods) arrived within approximately the same time, no large density peaks were observed, and deep-dwelling bivalves did not colonize in the first 2 years.

A diverse assemblage of fishes (7 to 9 spp.) immediately colonized the channels of both MN and TL in the first months after excavation (Fig 5.14). Total fish densities and species richness peaked in the first 5 to 17 months, with fish densities in the newly created channels far exceeding the range of densities found in local reference channels ($n = 3 – 4$) during these times. While the composition of fish assemblages at created sites was generally comparable to that at reference sites, relative composition showed an overwhelming dominance by opportunistic resident species (arrow gobies at MN, California killifish at TL). In both cases, species richness and density subsequently declined.

Restoration of fish and invertebrate assemblages depends on appropriate environmental factors, although the level at which fauna can discern constructed from natural sites may vary by taxon (Cammen 1976, Levin et al. 1996, Minello and Webb 1997). For many species, new sites offer resources that can be quickly tapped. For subsurface deposit feeders (e.g., oligochaetes), however, new sites may be uninhabitable due to the lack of food resources (e.g., insufficient organic matter in the substrate; Moy and Levin 1991; Levin et al. 1996).

Because most marsh restoration projects rely on natural recruitment, rather than deliberate introduction of animals, the rate and sequence of colonizers is influenced by reproductive rates and dispersal capability (Levin et al. 1996). Thus, the composition of faunal assemblages may change as site characteristics become more hospitable and as periodic reproduction makes individuals available.

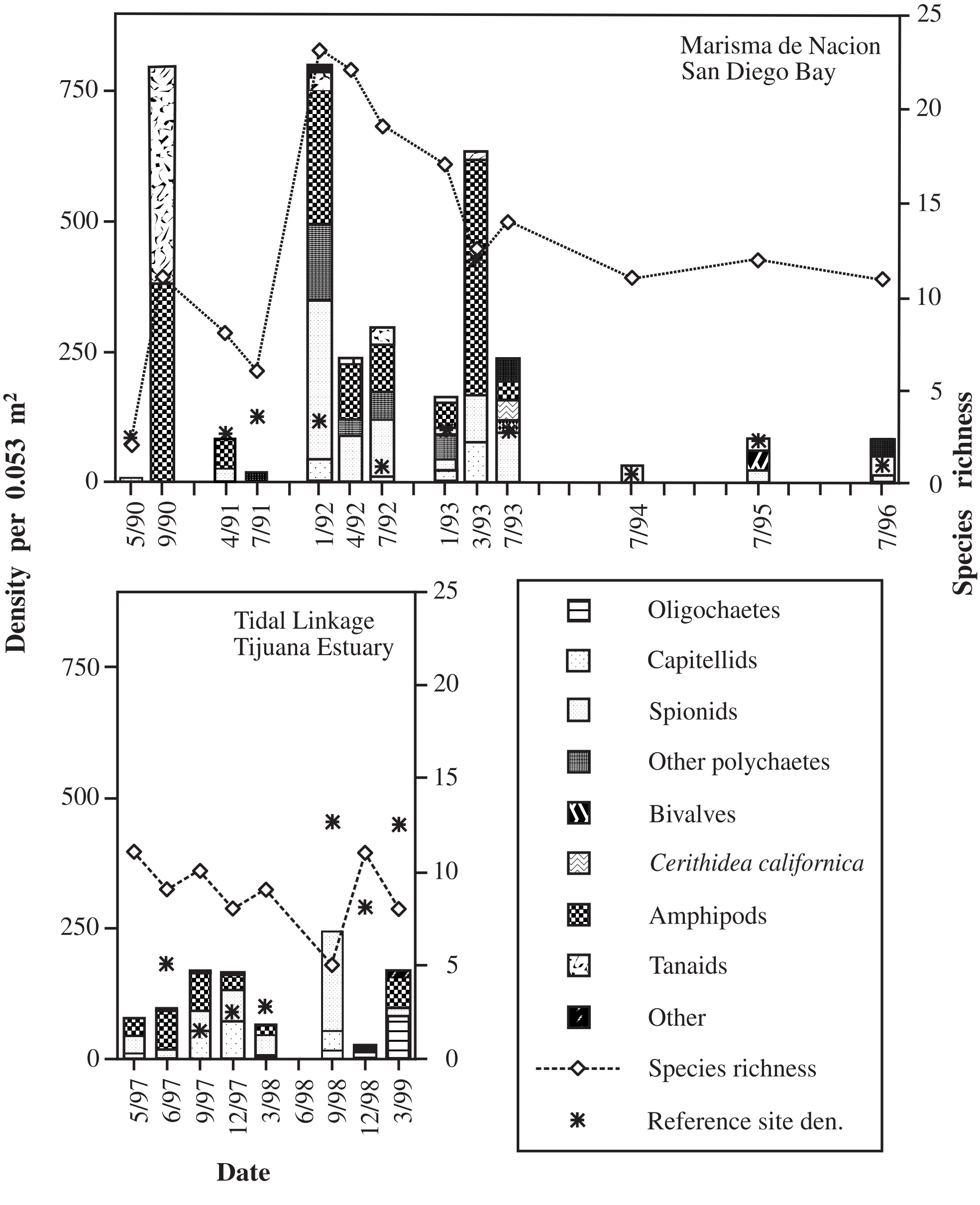

Figure 5.13 Invertebrate species densities (per 0.053 m^2) and species richness over time in channels of two constructed wetlands (Marisma de Nación and the Tidal Linkage), with natural reference site data indicated.

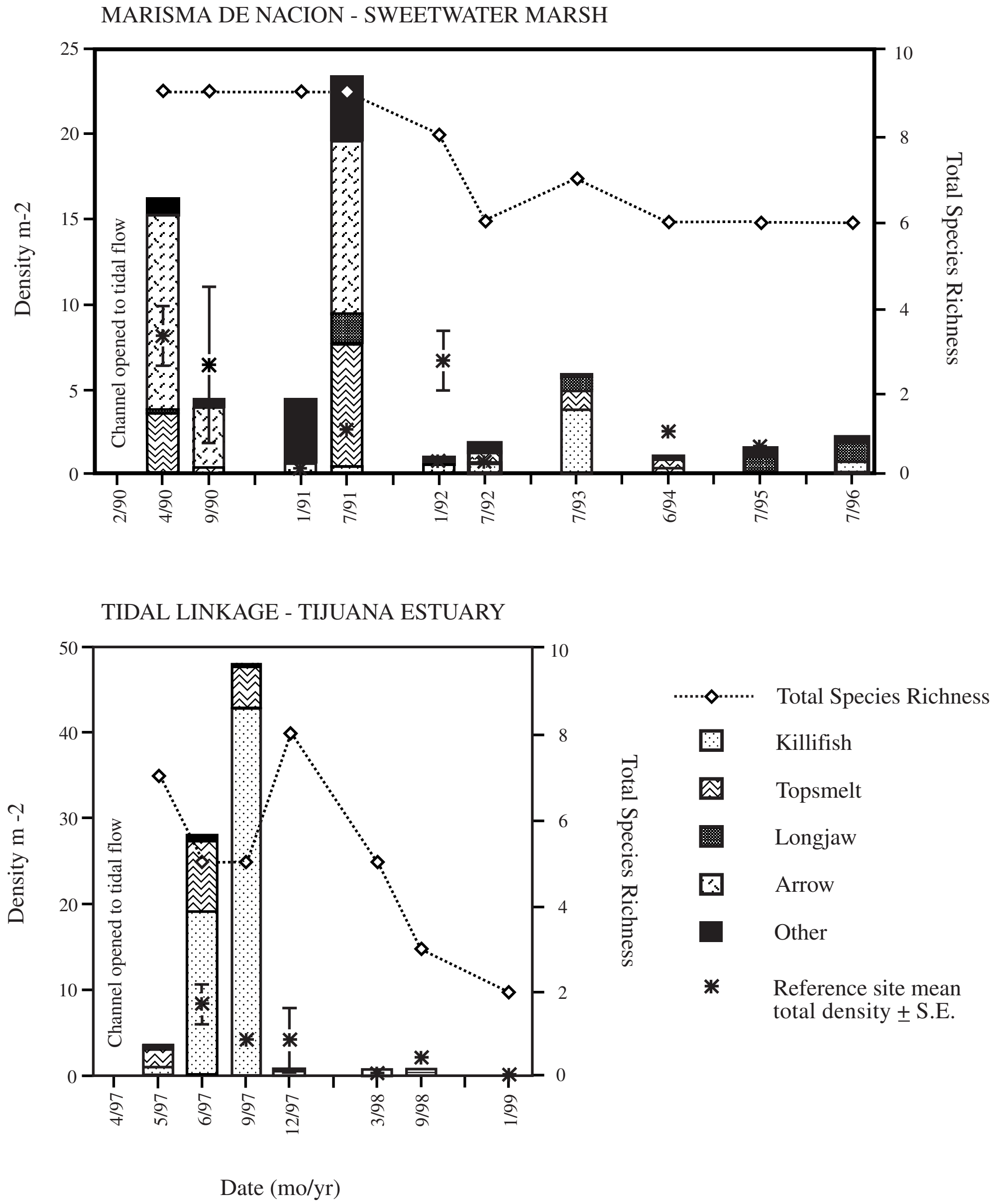

Figure 5.14 Fish species densites (per m^2) and species richness over time in channels of two constructed wetlands (Marisma de Nación and the Tidal Linkage), with natural reference site data indicated.

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chapter six

Assessment and monitoring

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6.l Introduction

Assessment is the quantitative evaluation of selected ecosystem attributes, and monitoring is the systematic repetition of the assessment process, that is, measurement of the same attributes in the same way, on a regular schedule. The placement and timing of samples is tailored to the spatial and temporal variability, including species' phenology and population dynamics. A one-time sample does not constitute monitoring, nor does the haphazard timing of repeated assessments or the repeated measurement of an attribute using different sampling methods. The essence of monitoring is consistency. At the same time, monitoring programs must be able to evolve. As data accumulate, it becomes possible to evaluate the monitoring program (Desmond et al., *in review*) using the knowledge gained to determine whether the monitoring can be streamlined, or to indicate where sampling may need to be increased and additional ecosystem attributes considered. Monitoring of restoration sites is best approached within an adaptive management framework, an iterative process that incorporates scientific findings into management decisions (Box 6.1). A good restoration monitoring program is one that (a) helps managers see how well the site is progressing toward restoration targets, (b) is conducted at spatial and temporal scales appropriate to the environmental and biological variability, and (c) is continually reevaluated.

Achieving a balance between consistency and adaptability is a challenge that requires experience with, and insight about, the ecosystem in question. Monitoring should assess the development of ecosystem structure and function, as well as address specific, pertinent questions. Simply collecting and cataloging data does little to advance the process of restoration or understand the site's development. To begin a monitoring program, it is better to over-sample, collecting data frequently from many sampling stations. An excess of sampling times and stations can easily be pruned after analyses suggest how often and in how many places each attribute needs to be assessed. We offer recommendations based on our experience with monitoring several southern California coastal wetlands, recognizing that our advice may not transfer verbatim to all regions.

6.1.1 Terminology

Throughout this book, we deliberately avoid the term "success" in characterizing restoration, construction, or enhancement projects. Rather than aiming for a one-word judgment, assessment and monitoring programs should be designed to (a) track the progress

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of ecosystem development and/or (b) determine compliance with mitigation requirements. If a site that complies with all mitigation criteria is considered a "successful project," then a project that meets all but one criterion would have to be called a "failure." It would be more informative to describe the site as "satisfying two out of three mitigation criteria." Furthermore, a mitigation project that met all criteria could still be a *restoration* failure if standards were poorly stated (e.g., requiring 80% vegetation cover without specifying "of native plants," allowing a site with 100% exotics to pass inspection) or if the site degrades after short-term compliance criteria have been met (as happened when the San Diego River changed its course and most of a "successful" riparian mitigation site reverted to upland conditions, J. Zedler, *personal observation*). By avoiding the term "success," we also avoid the term "failure." We prefer to describe the *degree of progress* toward functional equivalence with reference sites (Vivian-Smith discusses the need for reference sites in Chapter 2). Even if a site is judged "equivalent to" or "not distinguishable from" natural sites, the judgment should be qualified by indicating which attributes were examined. For example, one might conclude that, "at year 5, this site has made considerable progress, as it has vegetation coverage and native plant species lists comparable to those of three reference sites in this region, but exotic plants are still abundant in the high marsh." Some attributes will become similar to those of reference sites long before others (Craft et al. 1999). By substituting the word "progress," we recognize the complexity of the process of ecosystem development, and we try to avoid human bias (what is satisfactory to one person or group of individuals may not be acceptable to another). Hence, we strongly recommend that all evaluations of restoration sites clearly state the basis of the analysis and express the findings with that qualification.

Box 6.1 The adaptive management of San Diego Bay mitigation sites

Joy B. Zedler

The adaptive management of two mitigation sites in San Diego Bay (Connector Marsh, Box 1.7 and Marisma de Nación, Box 1.8) evolved over a 10-year period. The California Department of Transportation (Caltrans), the U.S. Army Corps of Engineers (CoE), the U.S. Fish and Wildlife Service (FWS), and the Pacific Estuarine Research Laboratory (PERL) each provided representatives to what became the adaptive management team. The team met annually to evaluate the results of ecosystem monitoring and to identify accomplishments and shortcomings of habitat designed for endangered species. The roles of each were as follows.

Caltrans and the CoE were the mitigators. Their projects (highway expansion and flood control channel construction) damaged wetland. Caltrans staff planned the marsh construction projects, and CoE contributed funding.

FWS was the resource agency and regulator; it determined that the damages to wetlands jeopardized habitat for three federally listed endangered species. Authority for setting mitigation requirements came from the Endangered Species Act. FWS set standards for compliance in 1988, following the 1988 settlement of a lawsuit, which reopened the Section 7 "Consultation" and allowed new targets to be set for the Connector Marsh (constructed in 1984 and planted in 1985) and Marisma de Nación (constructed in 1990;

opened to tidal flushing in February 1990, used for experimentation in 1990 ff., and planted in 1991). FWS's ongoing role was to determine if and when mitigation standards were met for each of the three jeopardized species.

PERL conducted monitoring under contract to Caltrans, solicited funds for restoration-related research, and conducted experiments on site with Caltrans' permission. PERL was joined by collaborators in the San Diego State University Geography Department who developed and implemented remote sensing of the restoration sites to assess the extent of different habitat types mentioned in the mitigation standards (Phinn et al. 1996).

PERL provided draft findings for Caltrans prior to each annual meeting of the team. This gave Caltrans an opportunity to decide if it wanted to suggest expanding or limiting monitoring. Caltrans needed to budget monitoring wisely, e.g., not sampling cordgrass for compliance until there was a chance that it would meet standards. Because the sites experienced insect outbreaks and other problems, vegetation assessment was delayed for several years. Research efforts (Langis et al. 1991, Gibson et al. 1994, and Zedler 1996*b*) helped identify the problems (coarse soil, low nitrogen supply, insect damage) and potential solutions (large-scale, frequent fertilization with urea). Once the site had matured, the 11.3 ha of constructed marsh were censused to determine if seven 0.8 to 1.6-ha areas, each with a variety of attributes, could serve as potential clapper rail home ranges.

Each year, PERL distributed a detailed report of both monitoring and research findings and made recommendations to the team about the status of the sites relative to the requirements. Early in the process, considerable time was spent determining exactly how generally worded standards should be defined (e.g., the requirement that cordgrass have "mean height of 60 cm" could be interpreted in various ways, depending on when, where, and how height was sampled). Hence, PERL included comparisons of sampling methods and recommendations of how to proceed. As team members became more familiar with the questions and cumulative data, the group became comfortable with the characterization of the sites, agreeing on conditions that did and did not meet the standards. FWS had veto power, but since the data were clear and strong, the critical designations (in compliance or not) were mostly developed through consensus.

The approach became adaptive once the monitoring and research results began to identify problems and suggestions from the field work were considered and implemented by the management team. The first site chosen for sowing seeds to reestablish the salt marsh bird's-beak produced plants but few flowers and seeds. Pollinators were a likely problem on the isolated marsh island. PERL suggested that the reintroduction site be moved to Sweetwater Marsh, a nearby, large remnant marsh with a larger upland buffer where pollinators might find additional habitat. The team gave permission for the effort to be moved; PERL seeded a much larger number of patches, and the population flourished. Monitoring for assessment began the year that the population numbered about 5000 individuals; in the next 2 years, the population was at about 14,000 individuals; the effort met (in fact, greatly exceeded) the requirement that at least five patches have at least 20 plants increasing in numbers or holding their numbers for 3 years.

The outcome of the monitoring was that standards for two species were met but nesting habitat standards for the light-footed clapper rail were not met, despite multiple attempts to improve habitat based on field research. Fertilizing and monitoring continued until the Connector Marsh was 13 years old. When we summarized our long-term data (Zedler and Callaway 1999), we predicted that soil nitrogen and organic matter concentrations would not match those of the reference site in a timely manner (>40 years). Furthermore, PERL identified ongoing habitat changes that did not suggest improvement in the short term, e.g., erosion of the marsh plain at Marisma de Nación (Haltiner et al. 1997) and cordgrass conversion to annual pickleweed at Connector Marsh (Boyer and Zedler 1999).

Both monitoring and research findings were critical to ending the mitigation program. PERL suggested that it would be futile to continue fertilization (it would only convert cordgrass to pickleweed). Instead, we suggested an alternative penalty (removing fill from another disturbed marsh). The team agreed that habitat for the light-footed clapper rail could not be provided in a timely manner, and that time and money would be better spent at another site. From a scientific perspective, however, the outcome was unfortunate, as it meant the end of a long-term data collection on one of the nation's most thoroughly evaluated habitat construction sites.

6.1.2 *Information needs*

Every restoration project poses specific information needs, and monitoring efforts will need to be tailored to provide that information: If tidal flushing needs to be restored, then monitoring will need to address water levels, current speeds, water quality, etc. If topography is to be altered, then elevations will need to be surveyed and evidence of erosion and sedimentation sought. If mineral soil is exposed and amendments are added, plants will need to be monitored for signs of nutrient stress so that amendments are adequate but not excessive. If specific plants are introduced, survival and reproduction will need to be monitored across the site to identify chances for their persistence. How much monitoring can be done is determined in large part by the funding available. When restoration is conducted as compensatory mitigation, monitoring will likely be geared toward compliance criteria, that is, determining if specific performance standards are being met. Once the site has complied, monitoring will likely cease (Box 6.1). Monitoring that is conducted outside the mitigation arena may have more general objectives and less intensive sampling (due to funding constraints), but potential for longer-term evaluation.

In this chapter, we describe and evaluate methods for assessing hydrology, soils, plants, and animals at restored wetlands. We introduce appropriate methods and refer readers to the literature for details on those methods. We treat the need for frequent sampling, and the recommendation that monitoring continue over the long term, both in reference and restoration sites (Box 6.2). In general, the more spatial variability within sites, the more sampling locations will be needed to characterize conditions. The more temporal variability, the longer a site will need to be monitored to plot its development path. Typical permits require 5 years of monitoring, but 10 to 20 years would be much more instructive. Ideally, monitoring should continue until the site is self-sustainable rather than ending at a fixed time limit. Although we recommend replicate sampling locations to incorporate the site's spatial variability, we do not provide a critique of sampling designs, that is, the number and placement of sampling units. Many books are available that discuss sampling designs for field experiments, which are relevant to restoration monitoring (e.g., Montgomery 1997, Elzinga et al. 1998, Krebs 1999), and various authors review statistical issues related to management (e.g., Peterman 1990, Cherry 1998).

A multi-purpose assessment program would satisfy the need to (a) plot the progress of a restoration site, (b) understand ecosystem development over time, and (c) determine when mitigation criteria have been met. For maximum ability to determine how restoration efforts have affected a site, one needs long-term data sets before and after restoration and several reference sites (controls) for comparison with the restored (or impact) site. This "before-after, control-impact" approach (BACI; Schmitt and Osenberg 1996) was designed to determine environmental impacts from large-scale, planned actions, such as the expansion of a power plant. The BACI design is also a useful approach for restoration

analysis (although "before data" are frequently missing for restored sites). In restoration cases, it is easy to show that the ecosystem undergoing rehabilitation is different from that of reference sites; what is more difficult is to determine when the restored system has become *similar enough* to be judged structurally and functionally equivalent to reference ecosystems or to be considered self-sustaining.

Cost, availability of interested and experienced personnel, and background knowledge on how best to sample (i.e., accounting for spatial and temporal variability for various taxa) will all influence the adequacy of the monitoring program. At San Diego Bay, we used our experience from other salt marshes to reduce the cost of monitoring for mitigation compliance. One mitigation requirement was that at least seven home ranges be provided for clapper rails, each with 100 m^2 of potential nesting habitat (tall cordgrass). It was thus unnecessary to sample cordgrass heights over the entire site. If the created marshes had been required to be uniformly tall and dense in cordgrass, we would have sampled sites randomly. Instead, we used remote sensing imagery to identify potential home ranges (attempting to satisfy several mitigation standards) and then chose sampling stations to represent the areas of densest, most luxuriant cordgrass and to characterize the best cordgrass that the site could produce. Later, we recognized that some readers of our publications assumed that we had sampled the average condition, rather than "the best" of the constructed site. Thus, our description of the constructed wetlands as "*less than* 60% functionally equivalent to the reference site," or "at best 60% equivalent," was interpreted as representing the entire site, when in fact the average condition was much worse. When this problem became apparent, we emphasized the unique nature of our comparisons and reiterated that our findings represent the best that constructed marsh had to offer, not the mean.

The San Diego Bay monitoring program (Box 6.1) fell short of the multi-purpose monitoring ideal because (a) monitoring was not commissioned until 1989, 5 years after the first mitigation work began, (b) monitoring was funded for only one reference site — the one damaged by construction projects (Box 6.2), (c) only the best cordgrass patches were evaluated, and (d) monitoring was terminated once we determined (in 1997) that mitigation efforts would not likely achieve compliance with mitigation criteria. Ideally, the resource agency that manages mitigation sites (the U.S. Fish and Wildlife Service, in the case of mitigation sites at the Sweetwater Marsh National Wildlife Refuge) would have adequate resources to fund long-term monitoring. However, at this point in time, no resource management agency has responsibility for determining the long-term progress of restoration or mitigation efforts. The Environmental Protection Agency (EPA) has commissioned the National Research Council to evaluate the policy and practice of mitigation, with a report expected early in 2001 (D. Policansky, *personal communication*).

Below are our recommendations for assessing the physical, chemical, and biological aspects of tidal wetlands.

Box 6.2 A single reference wetland at San Diego Bay

Joy Zedler

At San Diego Bay's Sweetwater Marsh National Wildlife Refuge (Box 1.6), one reference wetland, Paradise Creek Marsh, served as a target for the restoration of cordgrass (*Spartina foliosa*) marsh for nesting by the light-footed clapper rail. This marsh is one of only a few small remnants of salt marsh in San Diego Bay. Before highway construction, Paradise

Creek Marsh had supported 1 to 2 nesting pairs. A part of the marsh was damaged by highway widening, and the mitigation agreement required that lost habitat be replaced; hence the remaining marsh was the suitable target for mitigation efforts. While we would have preferred to use multiple reference sites, we were funded only to determine if habitat like that at Paradise Creek Marsh was being replaced. The mitigation agreement specified that constructed habitat had to measure up to that site.

Data from other coastal wetlands were sparse, but we used them whenever possible. Tijuana Estuary has a long-term monitoring program, now funded by the NOAA National Estuarine Research Reserve system, and we used sampling methods at San Diego Bay that allowed comparison with these reference data. Data from various wetlands were helpful in characterizing heights of cordgrass canopies that were and were not used for nesting by the rail (Zedler 1993*a*).

In retrospect, it was very important to have data collected simultaneously in constructed and natural marshes, as interannual variability was high. Only by being able to depict data from the constructed marsh relative to those for Paradise Creek for the same dates could we develop curves that showed marsh development (Zedler and Callaway 1999). In this way, the slow trend toward convergence of some attributes (TKN) and divergence of others (vegetation height) became clear.

6.2 *Hydrology and topography*

6.2.1 *Objectives*

Hydrologic and topographic surveys are used to characterize the tidal inundation regime and to evaluate the development of the site through changes in site morphology, erosion, and sedimentation. Monitoring these physical parameters is essential, because hydrology drives wetland development. Changes in all other ecosystem attributes require an understanding of the hydrological conditions.

6.2.2 *Inundation regime*

Remote dataloggers (e.g., Hydrolab, YSI, etc.) are easily used to measure depth and duration of tidal inundation for selected tidal cycles prior to wetland alteration and following construction of wetland habitats (Figure 6.1). By deploying multiple dataloggers across a site, tidal lags and the attenuation of tidal amplitude can be estimated. Measuring tidal amplitudes and lag times before and after restoration will indicate increases in tidal circulation due to excavation of new intertidal habitat. Placement of tidal staffs (simple, graduated measuring sticks) facilitates location of dataloggers at appropriate elevations.

Measurements should be made in both man-made and reference wetlands simultaneously to determine if the constructed system behaves similarly to a natural wetland. Measurements at increasing distances from the tidal inlet help to reveal any obstructions or bottlenecks in the main channels. The identification of a sill or hardpan that restricts tidal flows will help determine whether there is potential for increased tidal prism and whether such an obstruction needs to be removed.

To characterize the frequency and duration of tidal inundation, tidal heights are plotted with time over a 2-week tidal cycle for each location. On the California coast, the minimum tidal amplitude occurs in spring and fall, and the maximum in summer and winter; lowest tide levels occur in daytime in winter and at night in summer. Tidal regimes can be modified significantly within the estuary, with reduced amplitudes and more

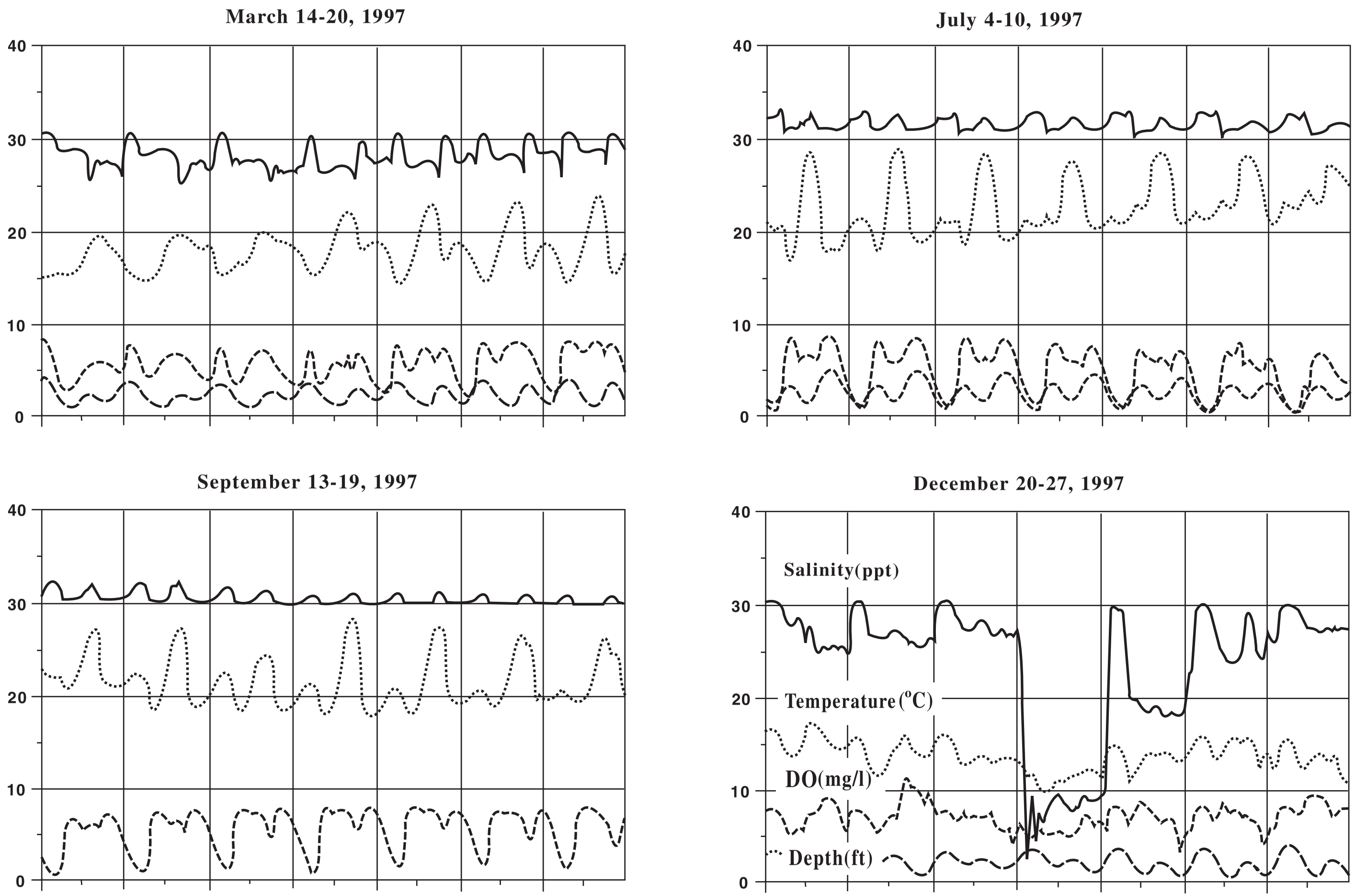

Figure 6.1 Water quality parameters measured from Tijuana Estuary using a YSI datalogger (PERL, *unpublished data*). Data are from one-week intervals during four sampling seasons and include salinity, temperature, dissolved oxygen, and water depth. Water depth data are not available for September 13–19, 1997.

delayed peaks at increasing distances from the ocean inlet. Van der Molen (1997) showed that the elevation of mean high water within Barnstable Harbour and Great Marshes in Cape Cod, Massachusetts, varied up to 0.65 m depending on the location within the estuary. Because of the large difference in tidal regime within an estuary, it is important that elevation be considered *only a general indicator* of tidal inundation.

Less costly than dataloggers are water collectors that can be deployed at known elevations across the marsh. Small vials are inserted into the soil surface along transects at locations that have already been surveyed for elevation. Prior to selected tides, vials are drained, and following the desired high tide, vials are monitored for water (Mark Page, *personal communication*). A simple tide gauge made of a dowel with a float can be used to record high water; a piece of folded plastic above the float is pushed up by the rising water, where it remains as a record of high water (Bill Streever, *personal communication*). The ground water recorder described below (Bragg et al. 1994) can also be modified to measured tidal inundation. Elevation of predicted and actual tides can be compared at multiple locations across natural and restored sites to evaluate tidal attenuation.

6.2.3 Ground water levels

In addition to measuring frequency of flooding by tidal inundation, it is useful to measure fluctuations in ground water, especially in the upper part of the marsh where tidal flooding may be infrequent but soil saturation affects plant distributions. Piezometers are simple lined wells that are used to measure ground water fluctuations (lining can be PVC or metal pipe, typically perforated at the appropriate sampling depth). They can be monitored either manually or with a datalogger. Faulkner et al. (1989) describe a piezometer that is useful for measuring shallow ground water levels in coastal wetlands. Piezometers have been used in a variety of studies to characterize ground water conditions at the wetland-upland transition for ecological research and wetland delineation purposes (Josselyn et al. 1990, Faulkner and Patrick 1992).

Bragg et al. (1994) developed a simple water level recorder that measures maximum and minimum water table depths. The float and housing are inexpensive and could be used at multiple sites across a restored wetland to monitor fluctuations in water table level. Additional field measurements and numerical modeling of soil water movements are described in Harvey et al. (1987) and Nuttle and Harvey (1995). An alternative measure of ground water levels is the steel rod oxidation method (Bridgham et al. 1991). Steel rods are inserted into the ground at known elevations across the marsh, and these rods are monitored for oxidation, as an indicator of soil wetness.

In addition to measuring fluctuations in ground water, it may be important to monitor ground water quality, especially for nutrient levels and salinity. In southern California and other areas with Mediterranean-type climates, moderate levels of ground water salinity combined with high rates of surface soil evaporation can lead to accumulation of extremely high concentrations of surface soil salts (Haltiner et al. 1997). In these cases, determining the source and quality of groundwater may be essential to maintaining suitable conditions for plant growth. Methods for collecting pore water samples for salinity and other water quality measures are described below (Section 6.4).

6.2.4 Flow rates

Water velocity measurements in tidal creeks are useful for understanding the geomorphological development of a site (Gordon et al. 1992, Haltiner et al. 1997), as well as for estimating the flux of water, nutrients, and sediments (Healey et al. 1981, Reed 1987, Reed

1988). Surface velocity is typically measured by timing the travel of a float over a known distance (Leopold et al. 1993), and this is sufficient for most analyses of geomorphology. For more accurate flux measurements, channel cross sections and depth profiles of water velocity are needed (Wang et al. 1994).

6.2.5 Creek development

High resolution aerial photographs or other remotely sensed images should be used to document changes in creek development throughout the restored wetland. These images are also useful for evaluating other large-scale changes in habitat distributions. Phinn et al. (1996) used multispectral digital video imagery (Airborne Data Acquisition and Registration [ADAR] 5000 sensor) to classify and map vegetation at a high spatial resolution. Desmond et al. (2000) also used ADAR to map distributions of creeks of different size and order in natural and restored wetlands. Historical maps and aerial photographs were used to document changes in geomorphology in marshes in Tomales Bay, California, dating back to 1862 (Niemi and Hall 1996). Large changes in marsh area were documented due to sedimentation in the late 1800s. Either aerial photographs or other remotely sensed images can be used to evaluate creek development, as long as the resolution of the image is sufficient. Large-scale monitoring should be combined with ground-based surveying and other fine-scale measures of creek morphology (Section 6.2.6).

6.2.6 Changes in marsh surface elevation: accretion and erosion

As a restoration site develops, it is likely to change in elevation (Section 6.2.5). The scale of changes may vary across the marsh, and methods depend on the magnitude of change in elevation. For large changes (>1 to 2 cm), surveying is useful, although small changes will require high precision and accuracy (Section 6.5.1). Surveying is particularly appropriate for creek cross sections, where changes are likely to be large, or in restoration areas that are designed to maximize accretion over the short term. For smaller-scale changes, marker horizons are advantageous, and sedimentation-erosion tables are useful in areas with high rates of subsidence. Geographic information systems can be used to evaluate spatial variability of accretion and erosion rates (Section 6.5.3). Each of these methods is discussed in more detail below.

6.2.6.1 Vertical reference points

Vertical positions are determined by measuring elevations relative to the National Geodetic Vertical Datum of 1929, i.e., the 1929 mean sea level based on multiple U.S. tide stations. Current mean sea level is about 10 cm higher than this. At present, the USGS is converting from NGVD to the North American Vertical Datum, 1988 (NAVD), resulting in some confusion in reporting reference elevations. In measuring and reporting elevations, it is essential to clearly state the datum used and to carefully relate elevations measured in the field to both a fixed datum and to the local tidal regime. In order to establish elevations relative to NGVD, a reliable benchmark must be located near the wetland of interest. Information on benchmark location and elevation is available from USGS, NOAA, and state and local agencies (e.g., planning departments). High precision and accuracy are needed in elevation surveys (Section 6.5.1). This represents a major challenge in remote wetlands and areas subject to subsidence where accurate vertical control is difficult.

In interpreting elevations across the wetland, it must be remembered that the inundation regimes of a specific elevation may differ for various locations within an estuary (van der Molen 1997). Tidal maxima and minima are attenuated at the inland extent of

tidal creeks. In addition, microtopographic features of the intertidal zone may impede inundation or drainage. Thus, elevation is a general indicator of inundation regime and vegetation. However, actual inundation at a site is affected by other factors beyond elevation, and predicted inundation rates should be compared to measured water levels throughout a wetland as described above (Section 6.2.2).

Changes in marsh topography can be evaluated by permanently marking transects or specific locations and resurveying these points over time (e.g., tidal creek cross sections). If elevational changes are greater than 1 to 2 cm/yr, surveying is very useful; however, smaller changes could not be detected reliably over the short term.

6.2.6.2 Marker horizons and sedimentation elevation tables (SETs)

Marker horizons, using feldspar or glitter, are used widely to measure sedimentation on the marsh surface (Harrison and Bloom 1977, Richard 1978, Cahoon and Turner 1989, Knaus and Van Gent 1989, Stoddart et al. 1989, Wood et al. 1989, French et al. 1995). Permanent stakes are also used, but stakes can increase local sediment accumulation, creating artifacts in the data. Marker horizons are the most effective method for measuring short-term accretion rates; they have also been used to track long-term sedimentation. This method works in areas that are consistently depositional but not where there are periods of erosion, such as creek bottoms, because the marker horizon can be removed. Nor are marker horizons useful in areas with high rates of bioturbation, such as mudflats. Bioturbation typically is not a problem in vegetated marshes, although fiddler crab burrows are extensive in some systems (Bertness 1985).

To measure accretion, a feldspar marker is laid down over the existing marsh surface, and the area is sampled at subsequent dates, using either a small coring tube (Cahoon and Turner 1989) or a cryogenic corer (Cahoon et al. 1996; Figure 6.2). The depth of newly deposited sediment on top of the marker horizon is measured, indicating the vertical accretion of sediment over the time period. The cryogenic corer minimizes compression and damage to the marker horizon, facilitating multiple corings and longer use. Measurements can be made directly in the field with the cryogenic corer. Sedimentation plates also can be used to measure short-term accretion rates. For this method, a thin, solid surface (often a plastic mesh) is placed on the marsh surface, and estimates of accretion rates are measured by inserting a calibrated rod into the sediment until it strikes the solid surface.

Sedimentation-erosion tables (SETs; Figure 6.3) offer a method of measuring marsh surface elevation with very high precision, e.g., within mm (Boumans and Day 1993, Cahoon et al. 1995). SET measurements are particularly useful in areas with high rates of subsidence (i.e., where changes in marsh surface elevation are affected significantly by more than just accretion) (Cahoon et al. 1995). A permanent pipe is set into the marsh to establish a benchmark, and the SET is attached to this benchmark to measure changes in the elevation of the marsh surface over time. Multiple pipes can be set into the marsh, and these stations are typically set up adjacent to marker horizon plots so that accretion can be measured simultaneously with changes in sediment elevation, allowing determination of shallow subsidence (Cahoon et al. 1995).

Reed (1989) developed a method using a filter paper to measure the mass sediment deposition for a series of tidal inundations. This method is labor intensive but offers accurate measurement of short-term sediment dynamics in restored wetlands. It has been employed in natural systems in conjunction with marker horizons to measure sediment processes on multiple time scales (French et al. 1995).

Long-term accretion measurements, such as ^{137}Cs and ^{210}Pb dating, are typically not useful for most restoration sites because of their young age; however, isotopic dating tools can establish patterns of sediment deposition in reference wetlands. Complications in comparing short- and long-term accretion rates are discussed in Callaway et al. (1996).

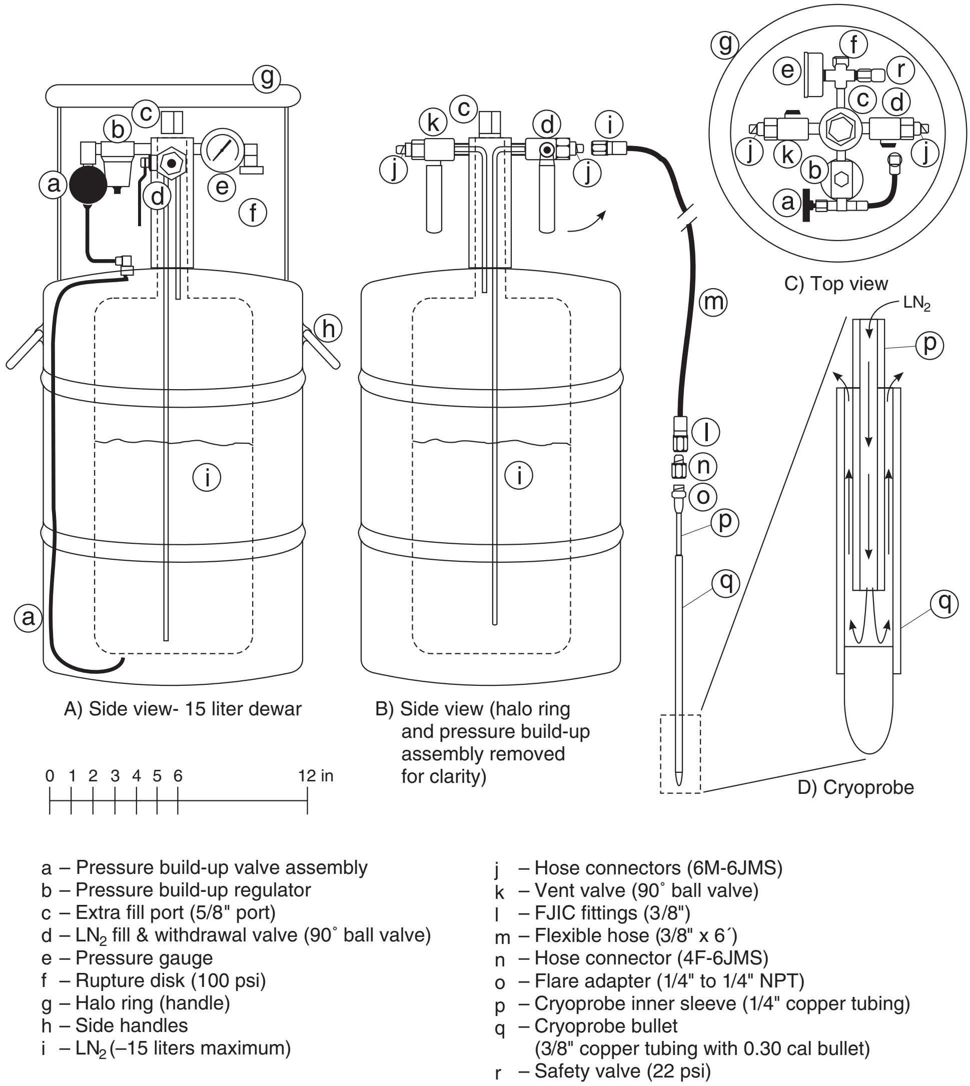

Figure 6.2 Cryogenic coring device for collecting soil samples to measure sediment accumulation with marker horizons. Soil freezes around cryoprobe, and compaction is minimized. (From Cahoon et al. 1996, Improved cryogenic coring device for sampling wetland soils. *Journal of Sedimentary Research* **66**:1025-1027, with permission.)

6.3 Water quality

6.3.1 Objectives

Water quality measurements indicate the physical and chemical conditions at a restored site. They are often indicators of poor circulation or impaired tidal flushing. As with other monitoring methods, sampling should be done across habitat types, including subtidal areas, intertidal flats, tidal creeks and channels, and brackish and freshwater ponds. Stations should be sampled biweekly, or at least monthly, to detect and account for seasonal as well as annual patterns.

Remote dataloggers allow detailed measurements on the scale of hours or minutes, which is valuable for parameters that vary widely during the day or week. Intensive

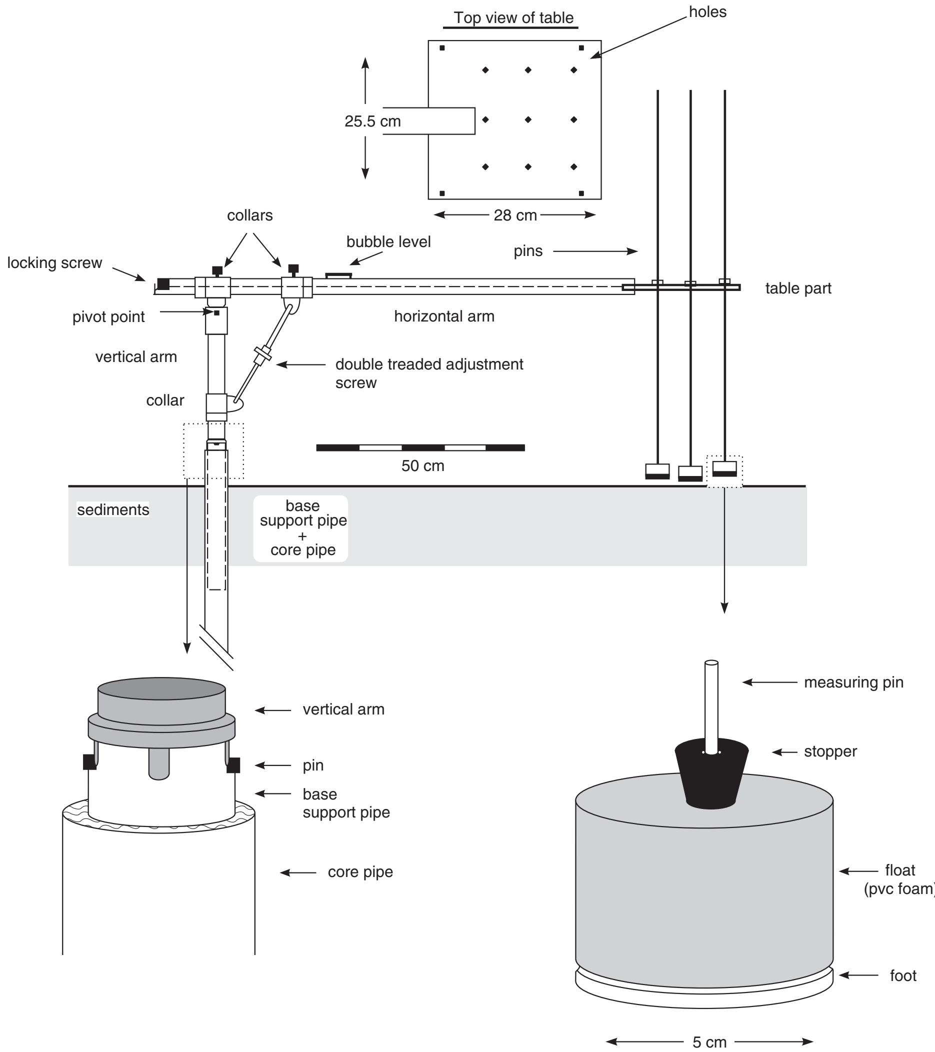

Figure 6.3 Sedimentation erosion table (SET) for measuring small-scale changes in marsh surface elevation. (From Boumans and Day 1993, High precision measurements of sediment elevation in shallow coastal areas using a sedimentaton-erosion table. *Estuaries* **16**:375-380, with permission.)

sampling with dataloggers can be very effective if it is targeted for particular events (e.g., following spring and neap tidal periods, during excessive freshwater inputs, during periods of likely tidal inlet closure, etc.). Dataloggers can measure most parameters of interest except for nutrients, including water temperature, dissolved oxygen (DO), salinity, pH, and turbidity.

Some analyses, including nutrient and algal measurements, require the collection of water samples. Water samples may need to be analyzed immediately or refrigerated, frozen or stabilized, depending on the information desired (Rowland and Grimshaw 1989, APHA et al. 1995). Water samples can be collected using a variety of standard methods. Surface samples are easily collected using any clean container. Subsurface samples require some type of remote system, with a sampling bottle, tubing, or some other collection device, depending on the stratification of sampling desired. Sampling methods are reviewed in textbooks on limnology and aquatic science (Lind 1979, Rowland and Grimshaw

1989). Balls and Laslett (1991) describe a sampler for heavy metal analysis that also would be useful for other water quality determinations. Readers seeking more detailed description of many of the methods below should consult references on water quality and chemical methods (Allen 1989, APHA et al. 1995).

6.3.2 Water temperature and dissolved oxygen

One-time measurements can give useful information for discrete events (e.g., runoff pulses or sewage spills); however, estuarine water quality varies greatly with tidal stage, time of day, storm condition, and season, so multiple measurements are needed to characterize general water quality conditions at a restoration site. Water temperature and DO are measured using a dissolved oxygen-temperature meter (e.g., Yellowsprings Instrument Model YSI 51B). Temperature data are particularly important in evaluating water column stratification. Temperature also directly affects metabolism and activity of many organisms and is useful in this regard.

Low DO levels cause stress and death for many estuarine organisms. To identify extremes and the duration of stressful temperature and DO conditions, we recommend using continuous sampling with a datalogger to record conditions during both spring and neap tides, over 24-hour periods. In intermittently tidal systems, the monitoring of DO following closure of tidal inlets or a reduction in tidal flushing is necessary to avoid killing of sensitive fish and invertebrate species (Chapter 5). Monitoring of this type has been very effective at Los Peñasquitos Lagoon (Box 1.4) to minimize impacts of low DO at this site, which intermittently closes to tidal action (Figure 6.4).

6.3.3 Water salinity and pH

Water salinity can be measured to the nearest part per thousand (ppt) using a salinity refractometer (American Optical, Reichert, Leica). Alternatively, a salinity/conductivity meter (YSI) and probe can be used. This instrument is preferred for sampling vertical salinity profiles because water collection is unnecessary. In addition, salinity/conductivity meters provide greater accuracy under brackish or low salinity conditions, where slight changes in salinity can have a substantial effect on vegetation. The refractometer reads salinity in ppt, whereas the salinity/conductivity meter can read in either ppt or conductivity units.

Water pH is measured in the field with a standard pH meter, calibrated to the field water temperature.

6.3.4 Light attenuation and turbidity

Light penetration can be measured using a submersible light meter (e.g., Li-Cor), sampling at the surface and at the bottom, with the depth between measurements recorded. Several different units can be used; the measurement of interest is % attenuation. The extinction coefficient is $k = [\log I_o - \log I_z]/z$, where I_o is the amount of light at the water surface and I_z is the amount of light at depth z. A simpler measure for deeper water bodies is to lower a Secchi disk (a 20-cm diameter disk) from the shady side of a boat or pier and determine the depth at which it is no longer visible. Secchi disk readings are simple and inexpensive, although this method is not as accurate as light meter readings.

Turbidimeters estimate turbidity based on nephelometric principles, measuring the amount of light scattered by a sample under controlled conditions (APHA et al. 1995). Turbidimeters give results in nephelometric turbidity units (NTU) and must be calibrated to solutions of known turbidity. Remote dataloggers also use nephelometric methods, but dataloggers can measure turbidity *in situ* using a turbidity probe.

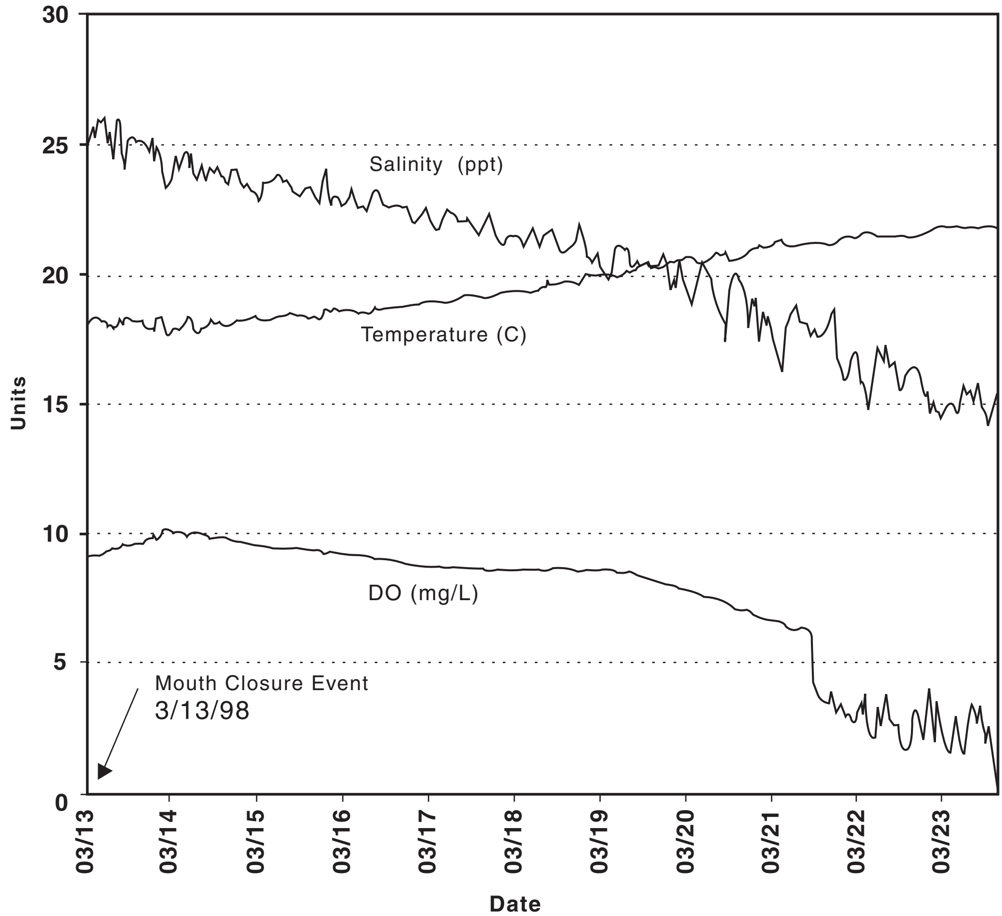

Figure 6.4 Dissolved oxygen, salinity, and temperature measurements from Los Peñasquitos Lagoon using a YSI datalogger (PERL, unpublished data). Note sharp decline in dissolved oxygen levels 8 days following the closure of the lagoon mouth.

6.3.5 *Water column stratification*

Impaired tidal flushing can be detected by monitoring the degree to which the water column is stratified (layered). In sluggish channels, water columns become stratified, with surface and bottom water differing in temperature and/or salinity. In winter, following rainfall and streamflow into the estuary, fresher (and possibly warmer) water may float over the more saline/cooler seawater. In late summer, the pattern of stratification may reverse, with warmer hypersaline water overlying seawater.

To characterize water column stratification, water salinities and temperatures are taken monthly at selected sampling stations. Temperature is first taken at the surface and at the bottom; if there are differences, additional samples are taken at 10-cm vertical intervals to locate the thermocline.

6.3.6 *Nutrients (inorganic N and P)*

Concentrations of nutrients in the water column are analyzed to assess eutrophication, which can lead to problems (algal blooms, Section 6.6.15) or benefits (improvement of

nutrient-poor marsh soils). Restoration sites can experience nutrient pulses when sediments are disturbed by dredging or following sewage spills. If, as described in Section 7.5, an algal bloom develops, new plantings can be damaged. If a nutrient pulse is detected, steps can be taken to deal with the algal bloom and any subsequent negative impacts (Section 7.3.2).

Nutrient concentrations are measured in water samples that are collected as described above. Tidal channels are usually well mixed, so sampling can take place at a uniform depth, pooling different depths, or pooling samples from several locations. We suggest pooling samples to obtain an average for the sampling station, followed by more detailed sampling if the average concentration is high, to determine the source of enrichment.

Samples can be frozen and analyzed at a later time, but more reliable measurements are obtained on fresh samples that have been kept on ice between the site and the laboratory. Water samples should be filtered first unless the attribute of interest includes particulates. Wet chemistry techniques are frequently used for nutrient analysis, although they are time consuming (Allen 1989, APHA et al. 1995). Routine analyses are commonly conducted on a spectrophotometer or autoanalyzer, which automates the process (Skoog and Leary 1992). Hach kits are also useful for some nutrient measurements. These inexpensive kits are widely used; however, they are less precise than wet chemistry or autoanalyzer methods. Samples that routinely exceed concentrations of 1 mg/l are suitable for analysis with the kits (US EPA 1993).

6.4 Soils: substrate qualities, nutrient dynamics

6.4.1 Objectives

Wetland soils develop fine texture and high organic matter content very slowly, and monitoring is important to identify constraints on plant growth. Although there is no standard reference for wetland soil methods, basic soil references are useful (Westerman 1990, Smith 1991, Council on Soil Testing and Plant Analysis 1992, Carter 1993).

High variance among samples is more often the rule than the exception for chemical parameters of soils. Variability within the sampling area can be reduced by collecting and compositing several samples before analysis (Binkley and Vitousek 1989). In general, one should increase replication as much as is practicable to improve the estimate of the mean. Replicate soil samples can be located along transects through different habitats or with randomly located sampling stations within each habitat. For sampling over time, it is necessary to stake sites permanently with rebar and surveying caps or other markers. Good markers will greatly reduce the time needed to relocate sampling sites. In areas with public access, there is a trade-off between easily marking sites and minimizing visual impacts or inviting tampering. It may be useful to obtain precise locations of sampling sites with a global positioning system to aid in their relocations (Section 6.5.2).

Analysis time is the usual limiting factor for soil chemistry studies. Compositing samples decreases variance without increasing analysis time. Lloyd and McKee (1983) have suggested a statistical procedure for determining the optimal number of subsamples required by the level of confidence desired. The number of samples will depend on the degree of variability of the measurements for the soil at a particular location.

Attributes that are subject to seasonal variation (water content, soil salinity, or nutrient concentrations) should be taken at least quarterly at the same tidal stage (e.g., low tide at the end of a neap period). Some measurements are best made more frequently (e.g., to document fluctuations in soil salinity over varying tidal cycles), while others change less often and can be sampled annually (e.g., soil organic content). Depending on the analysis of interest, soil and/or soil-water samples may be necessary. Each is discussed below.

6.4.2 Soil sample collection

Bulk soil samples can be collected with either soil probes or coring tubes. Probes that are 1 to 3 cm in diameter are useful for collecting small samples (suitable for soil salinity and total Kjeldahl nitrogen (TKN; Section 6.4.10)). However, it is difficult to collect soil samples to depths greater than 20 cm with a soil probe. Additionally, a probe must be pressed through the root zone, and the pressure compacts the soil samples. For larger samples, sharp-edged soil coring tubes (ranging in diameter from 5 to 15 cm) are very useful for collecting uncompacted samples for profiles of soil characteristics. Hargis and Twilley (1994*a*) describe a razor-sharp soil coring tube (Figure 6.5). Larger-diameter coring tubes cause less compaction, but they are more difficult to remove from the ground, are heavier, and cause greater disturbance to the wetland.

The depth sampled depends on the focus of the study. If the effect of soil salinity on seed germination is being assessed, one will likely want to collect samples from the soil surface (top 1 to 2 cm). When evaluating impacts on newly rooted plants, samples from 0 to 10 cm will be more appropriate. Mature plants may root to 50 cm depending on the species. In most cases, salt marsh plant roots are found in the top 20 to 30 cm.

6.4.2.1 Pore water collection

Several methods are available for collecting soil pore water. In choosing a method for a particular monitoring project, it is necessary to consider how much precision is needed in sampling by depth. For single samples where exposure to air is not a consideration, syringes, centrifuges, and vacuum-operated extractors are useful and easy. In cases where sites will be resampled over time but fine gradations in the depth of pore water samples are not needed, pore water wells are useful. When multiple depths are needed, or anaerobic conditions are necessary (as for some chemical analyses), dialysis samplers or "peepers" are most useful. Bufflap and Allen (1995) review some of these sampling methods and others for collecting pore water; their emphasis is on heavy metals, but the methods that are described are applicable for most pore-water sampling.

For the syringe, centrifuge, and airstone methods, a bulk sample is collected using the methods above, and pore water is extracted from this sample. These methods work best with relatively wet soils. For the syringe method, soil water is expressed by loading a 10 cc plastic syringe (without needle) with 2 layers of #2 Whatman filter paper that has been cut into 12-mm-diameter circles (punched from larger sheets using a half-inch die). The wet soil is loaded by hand; the plunger is inserted, and a drop of water is forced onto a refractometer (PERL 1990). It is possible to extract a few drops of water using this method, allowing salinity to be measured easily in the field.

Larger volumes of pore water can be collected with a whole-core squeezer, although this method is typically used with unvegetated sediments (Bufflap and Allen 1995). Centrifuging can also extract more soil water (Bufflap and Allen 1995). Depending on the need, samples can be filtered or unfiltered using the centrifuge method. Last, Winger and Laiser (1991) describe a simple vacuum-operated pore water extractor that uses a glass air stone to extract pore water samples. With a hand-operated vacuum pump, they were able to extract up to 1.5 L of pore water from 4 L of sediment.

When sampling is to be repeated, pore water can be collected conveniently by wells that are permanently installed in the soil. Wells should be sampled as soon as possible after a high tide when soils are saturated. To obtain a fresh sample, each well must be emptied with a syringe fitted with a vinyl tube (long enough to get to the bottom of the well) and allowed to refill before collecting the pore-water sample. Some parameters such as pH, redox potential, and salinity can be measured directly in the well, or samples can be taken into the laboratory for chemical analyses.

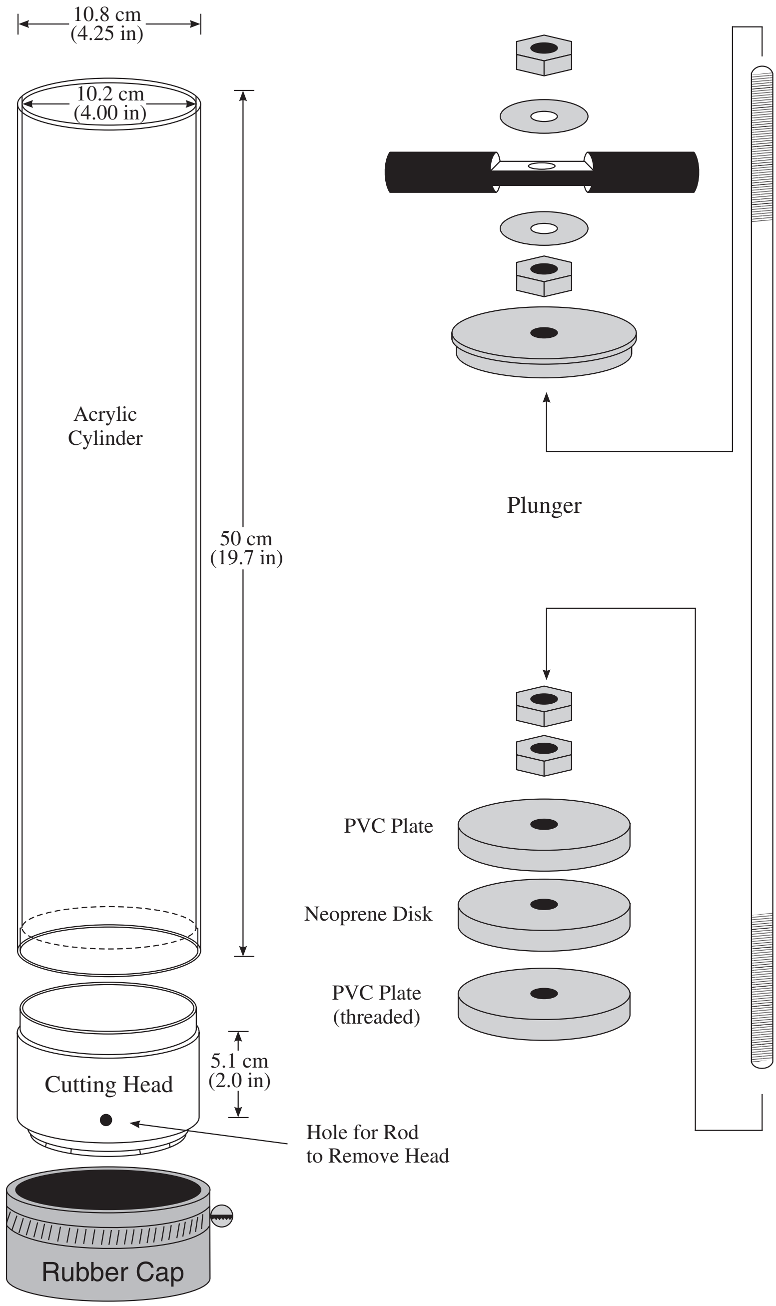

Figure 6.5 Coring apparatus for collecting uncompacted soil cores. Coring tube is shown as 10-cm diameter, although cores of different sizes are also available. (From Hargis and Twilley 1994*a*, Improved coring device for measuring soil bulk density in a Louisiana deltaic marsh. *Journal Of Sedimentary Research Section A: Sedimentary Petrology and Processes* **64**:681-683, with permission.)

Wells typically are constructed from plastic (PVC) pipes (2-cm inside diameter, 2.5-cm outside diameter), in which horizontal slits are cut at 2-cm intervals (depending on the desired sampling depth). The slits are covered with a piece of nylon screen held in place with a sleeve made from a thin PVC tube of dimensions similar to the well. Small particles

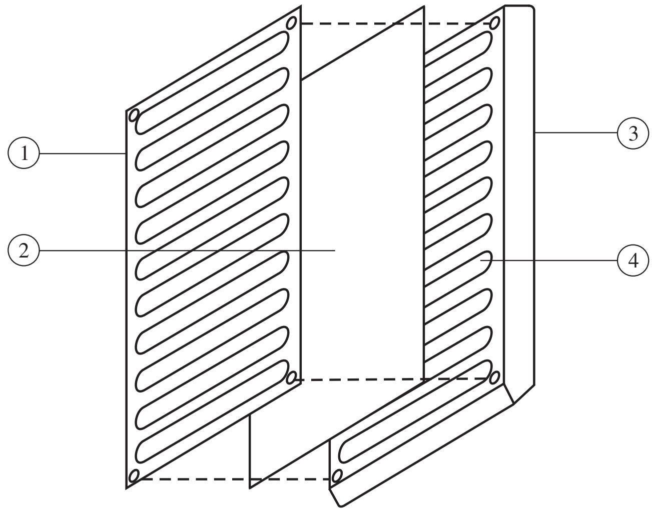

Figure 6.6 Pore water samplers or peepers for collecting pore water samples at multiple depths. Peeper sampling chambers are typically at 1 to 3 cm intervals. 1 = Plexiglas top; 2 = membrane; 3 = Plexiglas body; 4 = sample chamber. (Reprinted from *Water Research,* Volume 29, S. E. Bufflap and M. E. Allen, Sediment pore water collection methods for trace metal analysis: a review, pages 165-177, 1995, with permission from Elsevier Science.)

can move through the screen, so water samples should be allowed to settle. It might be necessary to centrifuge or filter the sample before analysis. The depth of sampling is adjusted by the location of the slits and by inserting the wells at different depths. It is possible to collect water from a narrow or broad range of depths using such wells. Craft et al. (1991*b*) used this type of well for extensive pore-water sampling at a restored wetland in North Carolina, as did Langis et al. (1991) in San Diego Bay.

Peepers are useful for sampling fine gradients in pore water conditions and when anaerobic sampling is necessary (e.g., with nitrate or metals). These samplers were designed by Hesslein (1976), and modified samplers are described by Bufflap and Allen (1995). The peepers are constructed of a plexiglass or PVC body, with holes or chambers machined into the body at set intervals to hold water (Figure 6.6). A cover with matching holes holds a dialysis membrane in place. The chambers are filled with DI water, and the peepers are placed in the sediment and allowed to equilibrate. Equilibration times from 3 to 20 days have been used (Bufflap and Allen 1995). Schipper and Reddy (1995) also used peepers for a study of plant decomposition.

6.4.3 *Water content (soil moisture)*

Water content is measured from bulk soil samples as weight loss upon oven drying divided by dry weight of the soil sample. The soil sample is dried to constant weight at 105°C (Gardner 1986). In tidal wetlands, soil moisture will vary dramatically depending on the tidal cycle. Samples should be collected at a standardized time during the tidal cycle (e.g., at low tide during a neap sequence) to minimize this variability.

6.4.4 *Bulk density*

Bulk density represents the dry weight of soil per unit volume. It is dependent upon soil texture and soil organic matter (Gosselink et al. 1984, Callaway et al. 1997*a*) and it is a useful indicator of soil compaction (e.g., from heavy equipment) and soil organic matter

content. Bulk density is estimated by weighing the dry mass for a sample of a known volume. Samples should be collected with minimal compaction (Section 6.4.2). The coring tube developed by Hargis and Twilley (1994*a*) is very useful for collecting soil bulk density samples. The soil is then sectioned by depth (so that a standard volume is sampled) and oven-dried to constant weight at 105°C. Bulk density = mass of soil/volume sampled; it is expressed as g dry-wt/cm^3 (Richards 1954).

6.4.5 Soil texture

Soil texture affects drainage, water and nutrient content, cation exchange capacity, organic matter accumulation, and many other soil properties. Texture is so important that samples should be taken during the earliest visits to potential restoration sites, in order to determine the need to import fine soils, as well as the potential for disposing of any substrate off site. Sandy substrate might be useful for beach replenishment. After the site is prepared, soil texture should be assessed again, and possibly after sedimentation events, to document the accumulation of new material. However, it is not necessary to measure this parameter frequently, as texture changes only with the accumulation of new material.

The particle size distribution of the soil is assessed in a slurry, in which particles gradually settle, allowing changes in specific gravity to be measured with a hydrometer (Gee and Bauder 1986). Simple formulas then allow calculation of sand, silt, and clay percentages. The detailed size differentiation methods used by marine ecologists (Emery settling tube and phi values) are not as useful for wetland soils, because high precision is not necessary. The hydrometer method requires 40 g dry soil. Sand particles remain in suspension very briefly, while clay will not settle in more than 2 hours. Soils collected for bulk density, soil moisture, or other sampling can be used for this analysis. Natural salt marshes typically fall in the clay to clay-loam categories.

6.4.6 Soil salinity

Soil salinity helps explain vegetation patterns and is useful in tracking the influence of freshwater inflows. Soil salinity is easy to measure in the field if the soil is saturated; otherwise, a saturated soil paste will need to be made, and results will approximate conditions after a rainfall. It is important to select a method that will allow comparison throughout the area of interest, which may include both wet creek banks and dry salt pannes. Sampling frequency depends on the question being asked. Soil salinity changes daily in the high marsh, in response to extreme high tides, rainfall, and evaporation.

It is important to measure soil salinity prior to planting vegetation at restoration sites. Dredged sediments and other materials that are used to create restoration sites (as well as conditions at some restoration sites such as lack of tidal flushing over long time periods) can be hypersaline. These conditions should be ameliorated by tidal flushing prior to planting. At Marisma de Nación in San Diego Bay, salinity of surface material was >80 ppt prior to the introduction of tidal water. PERL monitored conditions after successive tidal inundations and delayed planting until the soil salinity was under 60 ppt (Gibson et al. 1994).

If the marsh soil is saturated with water, the salinity measurement can be made immediately in the field, by expressing a drop of soil water onto a salinity refractometer using the syringe method described above (Section 6.4.2.1). If the soil is too dry, the sample is stored in a plastic bag and returned to the lab for artificial saturation with deionized water. The standard method for preparing saturated soil pastes should be followed; samples are wetted until saturation, with a key indicator being soil glistening (Richards 1954). Water from the saturation pastes is extracted using the syringe method and salinity measured with a refractometer. This method is time consuming, but soil samples can be

refrigerated and processed in batches. As an alternative, some researchers use a 1:1 (dry soil:water) soil solution.

Measurements of soils at their field wetness may give the most ecologically meaningful data; however, for soils with variable degrees of moisture (as is the case in most wetlands in Mediterranean-type climates, or in areas that are infrequently flooded), measurements at field wetness may not always be possible. In these cases, it is preferable to use the saturation paste method, so that comparisons can be made across different sites and times at similar levels of soil moisture. Results under saturated conditions can be viewed as the soil salinity that would occur following rainfall or tidal inundation. Comparisons of salinity across a variety of moistures may be needed to compensate for effects due to varying texture or other soil parameters.

6.4.7 Soil pH

Soil pH can be measured directly into moist soil or pore water wells, using a pH meter and a combination electrode. If pore water wells are not used and soil conditions are too dry, they can be sealed in a plastic bag and returned to the laboratory where this measurement can be made on soil paste (Section 6.4.6). A pH mini-electrode has been developed by de Jong et al. (1988), and this would be useful in cases where fine-scale vertical gradation of pH conditions is necessary.

In most tidal wetlands, soil pH is likely to be close to neutral (Ponnamperuma 1972). However, soil pH can vary by as much as 2 units within a tidal cycle because of water infiltration or benthic biological activity (Wolaver et al. 1986). Low soil pH could be a major concern at restoration sites where soils are drained and exposed to air. Salt marsh soils can become extremely acidic following the oxidation of sulfides to sulfuric acids. This situation occurs when tidal inundation is stopped and leaching of sulfuric acid is impeded by a high clay content (Mitsch and Gosselink 1993). Values lower than pH 4 are detrimental to salt marsh plant establishment. Broome (1990) noted no survival of vegetation planted in soils of pH < 3.

6.4.8 Redox potential

Measurement of soil oxidation-reduction (redox) potential is important because it affects the biogeochemical cycles of nitrogen, sulfur, and other redox-sensitive elements, including many heavy metals. Redox potential is measured by inserting a redox electrode directly into the soil or pore water well; measurements must be made with a calomel or other reference electrode. Redox electrodes can be purchased, or sturdier, field-oriented probes can be made. Faulkner et al. (1989) provide a detailed description of electrode construction, field installation, and measurement (Figure 6.7). These electrodes have been used for a year or longer in the field (Breaux 1992). Cogger and Kennedy (1992) describe a similar redox electrode design that is also suitable for extended field use. Farrell et al. (1991) have designed an inexpensive reference electrode that is suitable for field use, and Hargis and Twilley (1994*b*) developed a probe that can be used for simultaneous measurements at multiple depths. Faulkner and Patrick (1992) also have developed oxygen-diffusion chambers that are useful for the measurement of soil oxygen concentrations in moderately reduced soils.

Redox measurements have been used extensively in helping to identify and delineate wetland boundaries (Josselyn et al. 1990, Cogger and Kennedy 1992, Faulkner and Patrick 1992, Megonigal et al. 1993). A high degree of spatial variability is possible with redox measurements, due to differences in soil microsite conditions, so multiple spatial replicates are needed, especially for initial assessments (Fessel 1994). Probes should be allowed to equilibrate for 24 hours, and permanently installed electrodes are likely to be more stable

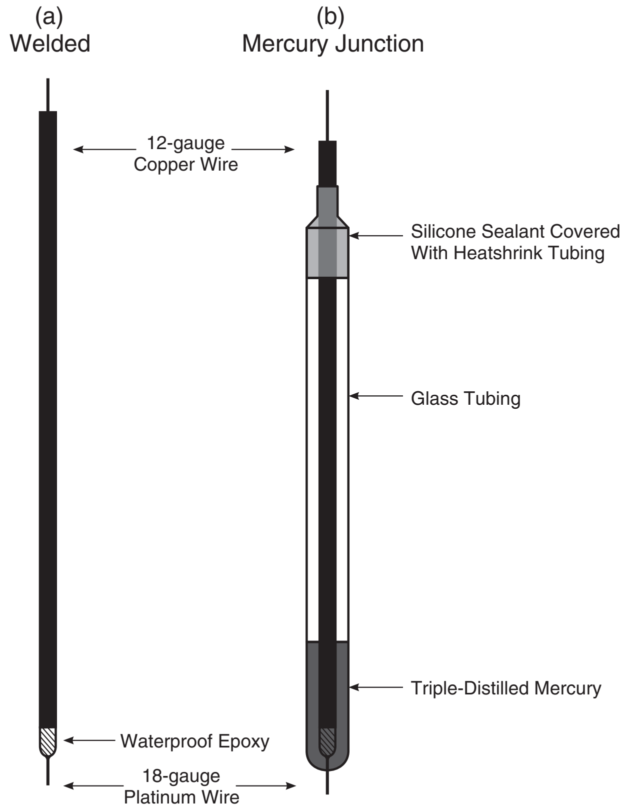

Figure 6.7 Welded (a) and mercury-junction (b) redox probes for measuring soil oxidation-reduction potential. (From Faulkner et al. 1989, Field techniques for measuring soil parameters, *Soil Science Society of America Journal* **53**:883-890, with permission).

than recently inserted electrodes (Steve Faulkner, *personal communication*). Equilibration times will be greater for seasonally inundated wetlands than for more regularly flooded areas. In addition, longer equilibration times will be necessary if electrode installation disturbs the soil.

6.4.9 Organic matter and organic carbon content

Both combustion and chemical oxidation methods have been used to measure soil organic matter (OM) or organic carbon (OC) content. Typically these are measured for bulk soil samples, although OC measurements are sometimes done on pore water. Combustion or weight "loss-on-ignition" is the most simple and common method for measuring wetland soil OM. For loss-on-ignition, dried and ground soil samples are burned in a muffle furnace, and the percent organic matter of the sample is measured as the relative weight loss after burning. A variety of burning temperatures and times have been used (Ball 1964, Davies 1974, Craft et al. 1991*a*). Burning at temperatures over 400°C overestimates OM when clay and/or carbonate content is high, because clays lose structural water and carbonates are destroyed at high temperatures (Mook and Hoskin 1982). Because of this, burning at 400°C for a longer period is preferred to higher temperatures for shorter intervals (Mook and Hoskin 1982).

Alternatively, OC can be assessed through wet digestion, and the results can be converted to OM content based on measured or standard ratios of OC/OM. The rapid

oxidation technique (Sims and Haby 1971) has provided satisfactory results. Oxidation in $K_2Cr_2O_7$ (potassium dichromate) is followed by treatment with H_2SO_4. Absorption of the diluted supernatant is measured on a spectrophotometer and compared to a series of standards. The main drawback of the method is that some of the refractory organic substances will resist digestion. Samples with significant refractory organic carbon are better measured by a second technique, the modified Mebius procedure (see Yeomans and Bremner 1988), where a more thorough oxidation is obtained by digesting the sample at 170°C with potassium dichromate, and OC content is measured by titration.

A final alternative is the use of a carbon analyzer (or a combined CHN analyzer). This method is appropriate for samples with very low or undetectable levels of inorganic carbon. If inorganic carbon is present at a significant level, it is advisable to use the modified Mebius procedure (Yeomans and Bremner 1988).

6.4.10 Nitrogen

The primary nitrogen pools of interest in marsh ecosystems are the inorganic nitrogen (NO_3^-, NO_2^-, and NH_4^+) that occurs in solution or is bound to soil and the organic nitrogen (living and dead material). Numerous texts provide in-depth discussions of biotically and abiotically driven nitrogen processes in the environment (e.g., Wetzel 1983, Salisbury and Ross 1985, Mitsch and Gosselink 1993). Nitrogen is frequently limiting in coastal wetlands, and most nitrogen in wetland soils is found in the organic form. Although measurements of inorganic nitrogen may be valuable in particular instances, soil total Kjeldahl nitrogen (TKN) is the single most useful measure of the nitrogen status of the soil. TKN measurements include the organic and inorganic forms (except for NO_3^- and NO_2^- forms). Nitrogen is measured as NH_4^+ after Kjeldahl digestion (APHA et al. 1995). The NH_4^+ content of the supernatant can be measured using wet chemistry or more typically with an autoanalyzer (APHA et al. 1995). Alternatively, total soil nitrogen can be measured with a CHN analyzer. It is desirable to assess soil nutrient concentrations before planting constructed sites to obtain baseline concentrations. Depending on the intensity of the monitoring effort, soil nitrogen may be monitored in both early- and late-season accumulation patterns (e.g., in August/September at peak aboveground biomass, and in December/January during dormancy for belowground reserves). If it is only measured once, sampling at peak biomass is recommended.

Measurements of NO_3^-, NO_2^-, and NH_4^+ estimate the nitrogen that is immediately available to plants. In the characteristically anaerobic sediments of salt marshes, most of the extractable nitrogen will be in the form of NH_4^+, with NO_3^- contributing very little to the nitrogen pool. These measurements can be made on pore water samples or extracts with KCl from fresh bulk soil samples. To prepare KCl extracts, place 10 g (wet wt) of fresh and homogenous soil sample in a 125-ml Erlenmeyer flask; add 50 ml 2 M KCl; and shake for 1 hour; filter through prerinsed Whatman™ no. 4 filters (the filtrate can be kept frozen until analysis). The filtrate is measured for NO_3^-, NO_2^-, and NH_4^+ using standard procedures (APHA et al. 1995).

Three nitrogen processes — nitrogen mineralization, nitrogen fixation, and denitrification — are useful to measure at restoration sites in order to determine why nitrogen might be in short supply. Estimates of nitrogen mineralization rates in the field can be obtained with the buried polyethylene bag technique (Eno 1960). Because plant uptake and leaching of nitrogen is prevented, net mineralization rates can be estimated as the change in concentrations of NO_3^-, NO_2^-, and NH_4^+ over time (Pastor et al. 1984) using cores placed in polyethylene bags in the field. White and Howes (1994) have also used isotopic methods to estimate mineralization rates.

Nitrogen fixation could be an important source of nitrogen for salt marsh vegetation. The most used and most straightforward method involves the acetylene reduction reaction, where the enzyme dinitrogenase is capable of reducing C_2H_2 as well as dinitrogen (Hardy et al. 1968, Casselman et al. 1981). In this technique, C_2H_2 is added to incubation vessels and dinitrogenase activity is monitored. Although this method must be calibrated with an ^{15}N tracer to obtain absolute values, relative comparisons can be made on the basis of n moles C_2H_2 reduced per unit soil (Zalejko 1989, PERL 1990, Langis et al. 1991). Piehler et al. (1998) used the acetylene reduction method and found higher rates of N_2 fixation in newly restored marshes compared to natural sites and concluded that this may have an important effect on the overall nitrogen budget of restored marshes.

Inorganic nitrogen can be removed from wetland soils and the water column via denitrification, and rates of denitrification are highest in soils with a dense rhizosphere and a large interface of aerobic and anaerobic zones (Reddy et al. 1989). Denitrification is measured using ^{15}N, acetylene inhibition, and N_2 flux. Seitzinger et al. (1993) compared these three methods and found that the acetylene inhibition method underestimates rates of denitrification, in part because acetylene inhibits nitrification and interferes with coupled nitrification-denitrification.

6.4.11 Phosphorus

Phosphorus levels are rarely limiting to plant growth in tidal wetlands, although inland wetlands often become eutrophic due to phosphorus pollution. Because phosphorus exists in a variety of forms, both total phosphorus concentration and phosphorus speciation are of interest. Total phosphorus can be measured following percholoric acid digestion (APHA et al. 1995). Several speciation methods have been used to determine the relative availability of different phosphorus fractions to plants. Extractable forms, in order from most easily available to least available, are: (1) labile or exchangeable phosphorus (iron and aluminum bound); (2) reductant-soluble phosphorus; and (3) calcium-bound phosphorus. Residual or bound phosphorus (mostly organic phosphorus) is usually determined by subtraction of the above available fractions from the total phosphorus concentration.

Methods for speciation are outlined in APHA et al. (1995). Poach and Faulkner (1998) used these methods to evaluate the availability of phosphorus in natural and restored wetlands of the same age in the Atchafalaya Delta. Craft et al. (1988) examined phosphorus pools and Craft et al. (1991*b*) measured pore water phosphorus concentrations in natural and restored wetlands in North Carolina. Craft and Richardson (1998) measured total P accumulation in Everglades wetlands, using ^{137}Cs, ^{210}Pb, and^{14}C dating.

6.4.12 Decomposition

Plant decomposition rates in soil have been measured in wetlands with litter bags (Van der Valk and Attiwill 1983, Schubauer and Hopkinson 1984, Hemminga et al. 1988, Hemminga et al. 1991, Rowe and McNichol 1991) or cotton strips (Hemminga et al. 1991, Day 1995, Conn and Day 1996). Both methods can be used to assess decomposition aboveground or in the soil, and both have been used to obtain decomposition rates over periods up to 5 years.

For the bag method, preweighed samples of roots and rhizomes or other macroorganic matter are buried in nylon mesh bags at specified depths. Initial samples are weighed wet because drying material affects decomposition rates. Additional subsamples are dried to determine a wet/dry ratio for the initial material. After a specific time period (anywhere from 1 month to multiple years), samples are excavated, rinsed, dried, and reweighed, and the percent weight loss is calculated. Problems with this method include

root ingrowth into bags and subsequent increase in biomass in the bags, difficulty rinsing material from partially decomposed roots, and variability in measured rates due to the heterogeneity of initial sample material. Because of these problems, some researchers prefer to use cotton strips to measure relative decomposition rates.

Cotton strips are buried in the soil, and after removal at specific intervals the relative integrity of the strips is measured by the loss in fiber tensile strength using a tensiometer (Maltby 1988). Strips produce more uniform results than litter bags; however, cotton is not native to salt marshes, so absolute rates of decomposition are not measured. The decomposition of cotton strips yields relative estimates, and this technique is useful for comparing constructed and natural marshes, as well as different sites within wetlands.

6.5 Elevation, global positioning systems, and geographic information systems

6.5.1 Determining elevation

Elevation can be determined through a number of methods. Standard surveying has been the most common method for determining elevation; however, laser levels and global positioning systems are being used more frequently to determine elevation and horizontal position of coastal wetland habitats. High resolution topographic maps can be used to approximate elevations for many applications, such as locating field sampling stations. Collins et al. (1987) completed a detailed, high precision survey (±3 mm) of the geomorphology of Petaluma Marsh, California, using an auto-level. Zedler et al. (1999) surveyed San Quintin Marsh (Baja California Norte, Mexico) in detail to evaluate the effect of elevation on wetland plant distribution. For constructed wetlands, engineering drawings detail slopes, tidal creek networks, depressions, and mounds; these drawings should be available from the planners of the site or the agency overseeing the project. Once the site has been constructed, "as built" elevations should be surveyed. It is unwise to plant vegetation without knowing the actual elevation, as sites are not always contoured as drawn on the plans.

Typically, an automatic level (e.g., Wild™ or Sokkia™ Instruments) and calibrated stadia rods are used to measure elevations at the site. Relative elevations are corrected to actual elevations by surveying to established benchmarks, such as a U.S. Geological Survey marker. Elevations are typically given relative to NGVD (National Geodetic Vertical Datum; see Section 6.2.6.1), although other reference data are also used (e.g., mean lower low water). In some cases, benchmarks may not be available and only relative elevations can be used. Surveying accuracy will be limited by the quality and resolution of the equipment and the distance between sampling locations. Sampling stations must be within line-of-sight, although this may be overcome by moving the auto-level to a different location, then backshooting the stadia rod from the new location and correcting for the difference. Laser levels offer extreme accuracy and precision in determining elevation and horizontal location over long distances, some over distances of 1 km or more.

Total station surveying systems can be used to determine locations and elevations at near or distant points with high accuracy (Figure 6.8). The total station itself is a combination electronic transit and distance-measuring device. As with a good auto-level and measuring-tape, one may determine the angle and distance from the instrument to each survey point. Distance is measured by shooting an infrared beam at a reflective surface at each sampling point. A reflecting prism mounted on a rod plumbed to the survey point is normally used as the target. Distance is calculated from the time the infrared beam travels to the target and back. Angles for triangulating coordinates are measured internally.

Figure 6.8 A total station surveying system is used to measure elevations at San Quintín Bay marsh in Baja California, Mexico.

The digital output is more accurate and less prone to recording or interpolation errors. The data can be downloaded and used to calculate relative surface coordinates in northing, easting, and elevation. Total stations are ideal for laying out plots or relocating points of interest on subsequent trips (e.g., marsh features, plots, or individual plants).

6.5.2 Global positioning systems (GPS)

GPS surveys provide high resolution data for the three-dimensional location of points in space: x-y-z coordinates on the earth's surface. By triangulating with a network of satellites, latitude, longitude, and elevation are measured to give the geographic coordinates of any point. GPS and line-of-sight equipment can be combined to provide an efficient and relatively error-free strategy for intensive surveying. GPS data can be transferred easily to geographic information systems with very accurate horizontal location of sampling points (Cornelius, 1994). GPS data are not as accurate vertically as fine-scale surveying or laser levels, but newer equipment offers ±1-cm accuracy. Repeated surveys can determine changes in key marsh characteristics, including species occurrence patterns, elevation ranges, expansion or decline in plant patch size, tidal creek network topography, channel erosion or migration patterns, sediment accretion rates, and/or tidal height dynamics over the marsh surface (Figure 6.9). For example, GPS data have been used to measure subsidence rates in the Sacramento Valley, comparing elevations over time as well as evaluating long-term change based on historic elevations (Ikehara 1994).

Figure 6.9 GPS technology is used to monitor the status of a rare plant, *Lasthenia glabrata* ssp. *coulteri* at Los Peñasquitos Lagoon.

High-resolution positional accuracy can be achieved through real time kinematic surveys with L1 carrier phase GPS receivers. Kinematic simply refers to the monitoring of carrier phase radio wavebands transmitted from the satellite network. A surveyor positions one receiver at a benchmark (the base station), with one or more others (rovers) taking field measurements. Field coordinates are calculated as position relative to the base station, determined from the satellite signals. The quality and resolution of the data depend on the sophistication of the equipment. Kinematic data are acquired instantly, so that many field points can be rapidly located and translated into coordinates (although accuracy may be enhanced with longer readings). Equipment without the benefit of "real time" correction takes 20 seconds or more to acquire each point, plus post-processing time to convert information into geographic coordinates (Zedler et al. 1997, B. Nyden *personal communication*). With real time kinematic surveys, maximum accuracy approaches 1 cm in latitude/longitude and 1 to 2 cm in elevation (Sokkia, Inc., representatives, *personal communication*). Although pocket-sized portable GPS receivers are useful for many applications, their accuracy is generally limited to approximately 100 m.

6.5.3 *Geographic information systems (GIS)*

A GIS is a computerized database that allows the user to combine mapped information from multiple sources, each providing a different type of information. The resulting digitized, georeferenced images are useful for storing, displaying, and using monitoring data (Lang 1998). Map sources originating in different coordinate systems, scales, or geographic projections (e.g., universal transverse mercator vs. state plane) are relativized to the same coordinate system and base map, using software such as Arc/Info™ or ArcView™ (ESRI 1997). The data "layers" (map overlays) may then be viewed in various combinations to determine relationships among them. For example, a GIS with separate maps of the distribution of a hemiparasite (e.g., salt marsh bird's-beak), the distribution

of its host species, the marsh surface topography, and sediment moisture levels can be displayed as overlaying layers. The user can then relate the distribution and aerial coverage of hemiparasite and host, elevation, proximity to channels, and sediment moisture. In subsequent years, information can be added on the location of plants, disturbance locations, foot paths, or land ownership boundaries. The resulting GIS becomes a very powerful tool for tracking and evaluating change and development in created or natural wetlands.

6.6 Vegetation

6.6.1 Introduction

Monitoring vegetation is perhaps the most effective means of assessing the development of restored salt marsh ecosystems. Periodic assessment of the plant community provides the basis for determining how restored systems are functioning relative to reference systems. Monitoring is also required to ascertain when predetermined benchmarks are achieved, indicating that functional goals have been met. However, wetland functional attributes are notoriously difficult to measure (Zedler 1996*a*), so monitoring programs primarily focus on structural attributes, such as the distribution and diversity of plant communities. These surrogate characteristics are easier and less expensive to quantify.

The monitoring strategy that is developed will depend upon the project goals. A common, overall objective is to establish wetland functions within a 5-year time frame. Specific goals may include establishing habitat for target species to mitigate the loss of habitat elsewhere. Other goals may be to provide minimum vegetation cover and/or transplant survivorship. Additional criteria are generally developed on a site-specific basis, depending on the availability of funds, type of site, mandates of the sponsoring agency, and/or losses to be mitigated.

A significant problem in vegetation monitoring is attributing differences between restored and natural sites to the restoration process. If a restored site is sampled in a year with flooding and heavy sedimentation, there are likely to be significant differences in plant cover and mortality when compared with data from any reference site(s) sampled in non-flood years. Differences might not be apparent if reference sites are sampled in the same year. Hence, we reemphasize the need to sample reference systems simultaneously for comparison with the developing restoration site.

First, we discuss the relationship between vegetation and ecosystem function and, second, we describe approaches for monitoring plant community development.

6.6.2 Vegetation as an indicator of ecosystem functioning

A basic characterization of restoration sites involves mapping the distribution of vegetation, based on surveys with several attributes of vegetation commonly monitored as indicators of wetland function. Individual species are targeted for a variety of reasons, depending upon their presence, rarity, endangered status, ability to indicate overall ecosystem health, or their identification as a mitigation target. The performance of any one species cannot fully determine or define how an entire ecosystem is functioning. However, several measures can be used to approximate ecosystem health and assess project development.

A basic and important indicator of function is the wetland's ability to support a diversity of habitats and species (Ferren et al. 1995). Recent studies of non-wetlands indicate the importance of species diversity to a variety of ecosystem functions (Ehrlich 1993), suggesting losses in diversity translate into losses in function. In mesocosm studies, net primary productivity, canopy layering, productivity, and CO_2 flux all increased with numbers

of annual plant species (Naeem et al. 1994, 1995, 1996). In species-rich grasslands, drought recovery, plant cover, and soil nitrates increased, while herbivory, disease, and weed cover decreased with greater diversity (Tilman and Downing 1994, Tilman 1996*a*,*b*, Tilman et al. 1996). These studies underscore the importance of maintaining sufficient diversity in restored systems, especially in the relatively species-poor coastal habitats where the link between diversity and function should be strong (Vitousek and Hooper 1993).

The presence of species over time (total species list; cf. Table 4.1) is a simple but useful indicator of ecosystem function, with their loss or decline suggesting environmental stress. For example, diversity is being lost in southern California salt marshes subject to the impacts of coastal development, such as freshwater inflow, sedimentation, and exotic species invasion. Although simply monitoring changes in biodiversity is not sufficient for tracking ecosystem development, this strategy can provide strong evidence that a restored site is in need of intervention.

Key indicator species, those with strong ties to specific habitats (Zedler 1996*a*), can link population increase or decline to changes in function (Section 6.6.9). In southern California, *Spartina foliosa* is valued for its regional rarity, high primary production, and as nesting habitat for the federally listed endangered light-footed clapper rail (*Rallus longirostris levipes*; Section 1.3.1). Woven canopies over the clapper rail's nest shields eggs and hatchlings from solar insolation and aerial view by predators, while the tall stems prevent buoyant nests from floating away on the tide (Zedler 1993*a*). *S. foliosa* is found only in fully tidal salt marshes in San Diego County (Appendix 3); it declines when tidal flushing is impaired (Zedler et al. 1992). Coulter's goldfields (*Lasthenia glabrata coulteri*) is a rare species monitored at Los Peñasquitos Lagoon, where it is threatened by excess sedimentation associated with development in the watershed. The reintroduced population of salt marsh bird's-beak (*Cordylanthus maritimus ssp. maritimus*) has been monitored at Sweetwater Marsh National Wildlife Refuge as part of the San Diego Bay mitigation project; its population shrinks to small patches at lower elevations during dry years. *Salicornia bigelovii* is common on the marsh plain of southern California wetlands that have good tidal flushing, but it was nearly extirpated from Tijuana Estuary during the 8-month closure of the ocean inlet in 1984 (Zedler et al. 1992). In our recent census of San Diego County marshes (Appendix 3), *S. bigelovii* was found only where tidal inundation occurred regularly; these were also the salt marshes with the highest overall diversity. *Salicornia virginica* provides nesting habitat for Belding's Savannah sparrow (*Passerculus sandwichensis beldingii*), a state-listed endangered species (Powell 1993); it often becomes a monotypic dominant when tidal flushing is reduced or eliminated. Data on the distribution and abundance of such populations can be used to characterize the progress of developing ecosystems.

The establishment of cover is commonly monitored. However, cover alone can provide a misleading picture of ecosystem function. Even after 100% cover has been achieved, other critical attributes, such as canopy height, structural complexity, above- and belowground biomass, nitrogen reserves, and soil organic matter content might be well below the levels found in reference systems. The taller and more complex canopies needed by many invertebrate (Boyer and Zedler 1996, Hacker and Bertness 1996) and vertebrate species (Powell 1993, Zedler 1993*b*, Zedler 1996*a*) develop as the ecosystem acquires biomass and nutrient reserves (Inouye et al. 1987, Barbour et al. 1999). Consequently, community structure and function will be evaluated more accurately by assessing canopy architecture, species diversity and abundance, and above- and belowground biomass, and nitrogen pools (Noss 1990).

Monitoring should establish a baseline from which to measure change in selected community attributes and to ensure that restoration objectives are being met. Assessment

should be broad and frequent enough to detect problems in need of intervention, such as plant losses due to herbivore infestations (Section 7.4). Below we discuss in detail procedures and techniques that can be used to assess marsh vegetation development and persistence. These include:

- mapping the distribution of vegetation across elevations and habitats,
- utilizing remote sensing,
- assessing the development of canopy cover *and* architecture,
- estimating changes in species richness and abundance, and
- estimating above- and belowground biomass and nitrogen pools.

Unfortunately, we know too little of the relationships between structural attributes and functions, such as biomass accumulation, nutrient accumulation (Zedler et al. 1980, Onuf 1987), and resistance or resilience to disturbance (e.g., wrack deposition and inlet closure). Above- and belowground biomass accumulation standards have rarely been incorporated into monitoring plans because minimum levels have not been developed for different habitats or community types. The level of effort required to extract information on function often exceeds funding for monitoring. However, we still recommend that functional attributes be assessed and compared with those in reference wetlands, so that appropriate surrogates can be developed (see Section 6.5.5).

6.6.3 Mapping the vegetation

Post-construction mapping provides the baseline data for evaluating subsequent development and for comparisons with reference wetlands. Depending on the scale of the project, maps can be prepared from remotely sensed data (Bradshaw et al. 1996), aerial photographs or digital images (Curran 1985, Jensen 1996, Wilkie 1996), and/or field surveys. Spatial data from the vegetation mapping should include total marsh area plus the absolute and relative proportion of area occupied by different vegetation types, habitats, and unvegetated space. In southern California, mapping units should include cordgrass habitat, marsh plain vegetation, high marsh, salt panne, and bare space. The development or decline of vegetation can be correlated with landscape features or physical attributes such as sediment texture, moisture, or salinity levels. Representative subunits of each vegetation or habitat type in both constructed and reference wetlands can be selected for detailed sampling, to document change or make comparisons among plant communities.

6.6.4 Remote sensing

Remote sensing of the spectral reflectance of coastal habitats allows coverage of habitat at the regional scale (Gross et al. 1989, Green et al. 1996). New techniques are rapidly being developed and adapted for monitoring applications, such as mapping the extent and type of salt marsh vegetation and assessing changes in composition over time. Maps of vegetative cover also provide the basis for examining the spatial distribution of species or habitats on the ground, such as rare plant or bird nesting sites (Brewster et al. 1996, Nyden et al. 1996). Georeferenced images are also useful for assessing the structure, and even the productivity, of wetlands. The remote sensing information can then be incorporated into, or provide the basis for, a GIS (Brewster et al. 1996, Phinn et al. 1996).

Given sufficient monitoring resources, information can be derived from the analysis of spatial, spectral, and temporal resolution of the remotely sensed data. Spatial resolution is

a measure of ground resolution element and image size needed to distinguish landscape features (Zedler et al. 1997). Spectral resolution refers to the number and identity of electromagnetic bandwidths being measured. Different wavelengths may be used to discriminate among vegetation types or between vegetated and bare areas. Temporal resolution refers to the time scale in which seasonal, diurnal, or tidal data are being acquired (Phinn and Stow 1996 *a*,*b*). Verification or "ground truthing" of the vegetative assemblages or surface features is required at the time remote data are acquired. Although remotely sensed data are relatively expensive to collect and analyze, availability should increase greatly over the next decade (Johnson-Freese 1995).

6.6.5 Vegetation attributes to assess

The methods employed in long-term monitoring programs of natural wetlands, such as the one begun at Tijuana Estuary in 1979 (Zedler et al. 1992), serve as a template for vegetation sampling in restored and created wetland ecosystems elsewhere (PERL 1990). Permanent sampling stations were established to assess trends in the plant community, recording species presence for frequency of occurrence and estimating relative cover for each species. More intensive surveys on stem density and heights were undertaken for *Spartina foliosa*, a target species in regional restorations. These extensive data on the inter- and intra-annual dynamics of salt marsh vegetation now provide the basis for long-term adaptive management decisions at the estuary. Similar methods may be used to assess development and temporal change in created wetlands, including ephemeral or random impacts within localized areas.

Wetland communities are often sampled at peak seasonal biomass (typically late summer or early fall, e.g., August or September in southern California). However, biannual surveys may be needed to survey species that are less or not apparent at this time. It is necessary to census some species more frequently to determine reproductive output, germination, and establishment dynamics (e.g., endangered species or invasive exotics). Seasonal growth and/or reproductive peaks vary among species, ranging from winter/spring to summer/fall active periods (PERL 1990, Zedler 1996*b*). Many high marsh annual species have completed their life cycles by the end of June, so estimates of biomass or population size should be taken in May or earlier (see Appendix 2). Exotic annuals should also be located and sampled in spring, as many species are relatively short-lived and dead by midsummer. Estimates of reproductive output require knowledge of the phenology of individual species, while estimates of recruitment should be done following the variable winter or spring rains responsible for germination events.

6.6.6 Vegetation transects and quadrats

A stratified-random positioning of sampling units is often desirable, because sampling along transects makes the relocation of points or quadrats easy (relocation is often the most time-consuming aspect of monitoring). Permanent transects with random starting points can have regular spacing of sample points, simplifying relocation. The newer GPS equipment, however, can guide the field crew to known coordinates, thus facilitating the re-sampling of randomly-located stations.

Permanent or long-term transects can be located in representative habitats and in areas of particular interest, e.g., sites likely to suffer impacts, locations of species that are rare, endangered, or good indicators. High-resolution topographic maps or GIS layers (if available) can be used to locate and set the length of transects (Section 6.5.3). Determining coordinates with a GPS and incorporating them into a GIS provides a permanent record.

Transect length depends on topography, as well as specific applications or goals. To sample representative areas within a habitat type, multiple transects can run parallel to elevation contours at randomly chosen locations (PERL 1990). Transects can also be perpendicular to elevation contours to capture the full range and/or spatial dynamics of vegetation change across the marsh (e.g., Zedler et al. 1999). Running transects across features (e.g., tidal creeks, berms, depressions) within or among habitats may be useful for characterizing local heterogeneity, especially on the marsh plain.

Quadrats of uniform size and shape can be positioned at regular intervals along each randomly-located transect line, and presence of species recorded. Quadrats can be located at 5 to 10-m intervals in small sites (relatively intensive sampling), or further apart in larger sites (extensive sampling). Alternatively, quadrats can be located at randomly chosen points within regularly spaced transects. To position a transect randomly, a pair of randomly chosen coordinates (x,y) locates an east-west line (x) on a map and a north-south line (y), with the start of the transect at the intersection of the two lines (Krebs 1999). Transects can be marked with sturdy stakes and numbered with marine paint. If they protrude above the canopy, the stakes might attract raptors that prey upon marsh fauna. It is not prudent to place predator roosts in a marsh occupied by endangered prey species (PERL 1990).

An important consideration for data on frequency of occurrence is the choice of quadrat size (Barbour et al. 1999). More species are encountered within larger quadrats (Arrhenius 1921, Schoener 1974). Quarter-square meter quadrats have proven suitable for many salt marsh vegetation surveys, although different sizes may be appropriate depending upon the scale of heterogeneity anticipated or the goals of the monitoring plan. For example, smaller quadrats may undersample widely spaced individuals or clones, while larger quadrats may group species that only co-occur in adjacent microhabitats. Quadrat shape should also be considered carefully. Circular sampling frames have less edge per unit area, reducing the number of decisions about which plants are inside and outside the sampling unit. Oblong quadrats will usually reduce variance in patchy vegetation. Whatever quadrat is chosen, size and shape should be held constant throughout the monitoring program.

At Tijuana Estuary, 0.25-m^2 circular frames are large enough to include multiple species (and allow analysis of interspecific interactions), and they are not too big to slow sampling. Presence of species is recorded; plant cover is estimated by eye. In the cordgrass, stems are counted and individual heights measured in 0.10-m^2 circular frames, as larger sizes include more stems than needed to assess height distributions.

6.6.7 Plant cover

Cover is an estimate of the proportion of area shaded by vegetation when the sun is directly overhead. It is a basic measure for assessing the development of created wetlands. Both absolute and relative cover estimates are used to explore species' contributions to the canopy in created and natural wetlands, to compare microhabitats within wetlands, or to compare changes over time. Visual estimates, intercepts along lines, or intercepts of points are all used to assess plant cover (Barbour et al. 1999). Visual estimates are fast but subject to bias. Lines are next in speed, with 10-cm intervals useful for recording salt marsh vegetation cover. Points can be identified using a frame with intersecting strings or a pin frame in which rods are dropped through the canopy to record hits. The latter method is needed to assess layering. A recent innovation is to use a laser pointer instead of a rod, making the system more portable and flexible.

Total cover is the proportion of area covered with canopy, calculated as the number of intercepts or points censused minus the number of intercepts or points of bare ground. For simplicity, the term “point” will be used hereafter. Because species canopies may overlap in mixed assemblages, it is essential to record bare ground as a separate item; i.e., the sum of points for all species often exceeds 100% of the points sampled. Absolute cover for each species is calculated as the proportion of sample points hit by that species. Relative cover for each species is the proportion of all points hit by that species, a measure of the relative contribution of each species to the canopy. The relative cover of all species sums to 1.0 (× 100 for percent). These data can be used to assess the development of relative abundance and evenness for each species.

$$\text{Total cover} = \frac{\text{total number of points samples} - \text{bare ground intercepts}}{\text{total number of points samples}}$$

$$\text{Absolute cover for species A} = \frac{\text{number of points intercepting species A}}{\text{total number of points samples}}$$

$$\text{Relative cover for species A} = \frac{\text{number of points intercepting species A}}{\text{sum of the hits for all species}}$$

Line intercept cover is taken by tallying the intercepts for each species at each interval within a transect line segment. Resolution in cover estimates is a function of interval length, with shorter intervals providing greater resolution. At Tijuana Estuary, 20-m lines (meter tapes) were randomly located, with intercepts recorded within each 10-cm interval. The use of 10-cm intervals provided cover estimates differing little from those collected with 1.0-cm intervals (PERL, unpublished data). Interval length should be adapted to local species occurrence, with smaller intervals needed to document more fine-scale patterns.

Point-intercept data are collected with a pin frame (Smith 1996, Barbour et al. 1999). Frames are randomly located within each sampling site, so that the data collected from each frame represents a random sample for statistical comparisons. Pins are lowered through the canopy, and each canopy intercept (hit) is recorded for each species. Theoretically, pins have no cross-sectional area, although a variety of thin and stiff materials may be used. Brass welding rods that are 1-m long × 1-mm diameter work well in short canopies. Others have incorporated laser pointers into frames for rapid measures of the upper canopy, but this requires intrusive manipulation of the upper vegetation to reach lower strata (Richard Ambrose, *personal communication*). Cover calculations are made from the number of pins hitting canopy for each species and/or bare ground for each frame. For cover calculations, a pin hit is analogous to a line intercept. Pin frames may be relocated within each area for replicate sampling. Frame and sampling designs may vary widely and be modified for specific applications.

Point and line-intercept sampling are widely used to quantify species cover and frequency of occurrence (Smith 1996, Barbour et al. 1999). In a local comparison of the two methods, differences between random point intercepts and line intercepts taken at 1-cm intervals were not significant (Eng and Nordby 1998). The intensity of sampling (number of replicate quadrats, line segments, or point sets) can be adjusted to suit the species present and the monitoring goals.

Visual cover estimates rely on classes. Six classes are used in the Tijuana Estuary monitoring program (<1, 1 to 5, 6 to 25, 26 to 50, 51 to 75, and 76 to 100%). Other classes have been used to estimate cover, usually with narrow classes at the extremes and broader classes around 50% cover (Barbour et al. 1999). Estimates of cover should be calibrated

with measurements and precision should be determined by asking samplers to resample the same plots after a break, to obtain a measure of repeatability. Frequency histograms of cover classes may be compared among sites with Kolmogorov-Smirnov two-sample tests (PERL 1990). Overall cover has also been assessed by hand-held radiometry, a technique used effectively in mesocosm experiments at Tijuana Estuary (Callaway et al. 1997*b*).

6.6.8 Canopy architecture

No single measure can fully characterize canopy structure, especially when different morphologies are considered. The vertical structure or architecture of marsh vegetation can be a measure of habitat value for other species. Consequently, intensive analyses of canopy structure may be called for in restoration projects, with sampling strategies tailored for particular species or communities. Heights of stems are difficult to measure for pickleweed (*Salicornia virginica*) because of the decumbant and highly branched growth form. In contrast, cordgrass (*Spartina foliosa*) culms are easily distinguished and their heights are readily measured by extending leaves upward along a meter stick. Clapper rails nest where cordgrass stem density exceeds 100 stems m^{-2} and where at least 90 stems exceed 60 cm tall and at least 30 of these stem heights exceed 90 cm (Zedler 1993*a*). Heights sampled within 0.10-m^2 quadrats are summed to give total stem length (TSL in m/m^2; a surrogate measure of biomass; Gibson et al. 1994) and counted to give stem density (stems/0.1 m^2; Covin and Zedler 1988).

Canopy height, cover, and the number of canopy layers can be integrated for each species from pin frame and canopy height measurements to assess complex architecture. Canopy layering at each pin frame is calculated by summarizing the number of canopy hits per pin for each species. Each pin hit (plant intercept) represents a canopy layer. The number of hits per pin for each species is averaged to get the mean number of canopy layers. It is advisable to average only pins receiving at least one hit, because dividing by the total number of pins will underrepresent the architectural complexity of the vegetated areas if many pins are hits. The calculation method should be clearly indicated.

$$\text{Layering of species A} = \frac{\text{number of hits for species A per frame}}{\text{total number of pins with} \geq 1 \text{ hit per frame}}$$

Incorporating height data into the number of pin hits provides an estimate of layering density (hits cm^{-1}), perhaps a key factor from an invertebrate point of view.

$$\text{Layering density for species A} = \frac{\text{number of layers for species A per pin}}{\text{pin intercept height}}$$

Achieving habitat quality standards within 5 years is a common restoration or mitigation goal. Although full cover may develop within this time frame, the architectural complexity characteristic of natural wetlands may take many more years to develop (Keer 1999). Performance standards for canopy architecture have not yet been developed for marsh species other than cordgrass. Until the necessary research is accomplished, monitoring strategies should incorporate experimental comparisons with reference marshes.

6.6.9 Target species

Monitoring plans may require assessment of the presence or abundance of particular species that impact function or indicate habitat quality. Examples are invasive exotic species, such as rabbit-foot grass (*Polypogon monspeliensis*), or important indicator species

such as the endangered salt marsh bird's beak (*Cordylanthus maritimus ssp. maritimus*) or the rare Coulter's goldfields (*Lasthenia glabrata* ssp. *coulteri*). The latter are narrowly distributed annuals: salt marsh bird's-beak is restricted to specific areas within its host range, while Coulter's goldfields is found on the margins of salt pannes and vernal basins. In southern California, the growth and reproduction of these species may vary a great deal both within and among years, strongly influenced by the quantity and timing of rainfall. Consequently, such species need to be mapped and monitored for several years to characterize their population sizes. Patch size can be measured as an ellipse with maximum and minimum diameters. Individual counts can be made in randomly located quadrats of various sizes, depending upon patch size and density. For the reintroduced population of salt marsh bird's-beak at Sweetwater Marsh, we tested several subsampling techniques and ultimately decided that a total count was necessary because the population was so patchy. When the population exceeded 10,000 plants, we estimated numbers of the densest patches to give a total population estimate (Parsons and Zedler 1997, Fellows 1999, Noe 1999).

GPS equipment greatly aids mapping of species distributions and correlating aerial and elevation distributions with physiological or biotic factors that might restrict them to narrow or ephemeral environments (Skinner et al. 1995). Periodic surveys allow correlation of the temporal dynamics with variation in annual weather patterns or episodic events such as ENSO. Extensive sampling of covarying factors, such as density, co-occurring species and specific microhabitats, can be coded into each three-dimensional position and used as the basis for modeling population dynamics. For Coulter's goldfields at Los Peñasquitos Lagoon (Box 1.4), the size and location of each patch was defined by a polygon of border coordinates. Interannual variations within and among basins were strongly related to soil moisture and salinity (Noe 1999). This type of information is critical for understanding what limits species distributions, and that understanding is essential to designing future restoration sites so they can support more diverse assemblages and be self-sustaining.

6.6.10 Biomass and nitrogen pools of marsh primary producers

Primary productivity is the rate at which photosynthetic energy accumulates in an ecosystem. It is a fundamental system function and a basic indicator of performance. The energy of primary production is accumulated in plant biomass and represents the ultimate source of energy for all trophic interactions. This biomass is also directly or indirectly responsible for the trapping, accumulation, and storage of nutrients above- and belowground. Marsh vegetation can trap organic sediments passing through the system and absorb nutrients released in decomposition. Belowground organic material binds soil nutrients and prevents them from being leached out of the system. Primary production not only changes the chemical and physical character of the environment, but it becomes part of the environment in which other organisms live. In addition to salt marsh plants, benthic microflora, submersed angiosperms, macroalgae and phytoplankton also can contribute significantly to wetland productivity (Zedler 1980, Thom 1984, Onuf 1987).

6.6.11 Productivity estimates based on carbon fixation

Primary productivity is an important measure of ecosystem development. Development can be assessed by comparing created and natural wetlands or plotting productivity within the restored wetland over time. Productivity estimates describe the net energy gain (after respiratory costs), with each carbon atom fixed representing a minimum gain of 12 photons.

Carbon fixation is measured with an infrared gas analyzer as the rate of CO_2 uptake under given sets of conditions (Pearcy 1991). It is commonly measured in a chamber on a portion of plant material (e.g., a leaf, branch or small plant) to get mass specific instantaneous rates of carbon uptake or gain (μg/g/s). This method allows comparison of individual leaves and plants; it has the advantage of not requiring destructive sampling. Its disadvantage is that the rates of photosynthesis are snapshots in time; they are not readily usable for calculating annual productivity. When biomass-specific estimates are available, productivity can be integrated over a day, season, or year on an aerial basis and calculated as $J/m^2/yr$ (energy) or $g/m^2/yr$ (biomass). However, it is generally difficult to extrapolate from carbon fixation rates to net primary productivity (NPP) for the ecosystem because there are many sources and sinks that are unaccounted for. In addition to salt marsh plants, benthic microflora, submersed angiosperms, macroalgae, and phytoplankton all make significant contributions to wetland productivity (Zedler 1980, Thom 1984, Onuf 1987).

6.6.12 Standing crop and its use in estimating productivity

Because of the difficulties inherent in measuring and interpreting carbon fixation rates, it is easier to measure standing crop, or the amount of biomass gained on an aerial basis (g/m^2). Standing crop is a surrogate for carbon gain in that the number of carbon atoms in plant tissue correlates well with dry mass. Measuring standing crop requires collecting multiple above- or belowground samples in representative habitats for dry mass calculations. This integrates carbon gain throughout the time period prior to harvest. However, standing crop of southern California salt marshes is a poor predictor of NPP because of the many losses that occur between sampling times and the differences in the amounts and timing of losses for different species (Onuf 1987 and *personal communication*). Collecting representative samples of biomass in species-rich and/or heterogeneous habitats is extremely difficult. Even for monotypes, standing crop estimates do not account for losses due to tissue senescence, damage, or herbivory (Onuf 1987). For evergreen species such as *Salicornia virginica*, the values can be far less than that determined by tracking new growth (Onuf 1987 and *personal communication*).

Although standing crop measures are commonly taken to estimate production under field conditions, destructive sampling is not advised for vegetation that is in the early stages of development. Multiple sample collections may also cause significant disturbance at restoration sites or the small natural wetlands typically found in southern California. Multiple harvesting methods, such as the Smalley method, have been used widely in estimating aboveground productivity in coastal wetlands on the Atlantic and Gulf coasts (Hopkinson et al. 1978, Gallagher et al. 1980, Gross et al. 1991). Linthurst and Reimold (1978) evaluated five harvest methods at three wetlands on the Atlantic coast and found up to ten-fold variation in the estimates of net primary productivity among the methods used. These harvest methods are not recommended for most restoration wetlands, although they may be useful at larger sites or for assessments after 5 to 10 years of development. Consequently, investigations of standing crop should be limited to specific research needs rather than used routinely in monitoring programs (PERL 1990).

6.6.13 Nondestructive estimation of aboveground biomass

Surrogate measures of aboveground biomass are useful where destructive sampling is prohibited or excessively damaging (Section 6.6.12). Measurements such as plant height, stem density, cover, or layering (or a combination of these parameters) can be used as the surrogate. Total stem length works well for cordgrass; it correlates well with biomass, and

it is easily measured in 0.10-m^2 quadrats (Zedler 1983, Covin and Zedler 1988). Total stem length is a good measure for plants for which individuals are easily distinguished and where there is a good correlation with biomass (bushy plants might not qualify). To establish the correlation, one needs to devote small areas to destructive sampling, first measuring the surrogate, then collecting the aboveground biomass within sample quadrats and obtaining dry mass. The size and number of quadrats needed to provide an adequate sample will depend on the heterogeneity of the vegetation.

Single estimates of peak biomass can be made, or multiple estimates of biomass over time can be used with the various methods for converting changes in biomass into annual productivity estimates (see Linthurst and Reimold 1978 for a review of productivity estimates). Morris and Haskin (1990) used an allometric method to estimate biomass of *Spartina alterniflora* and to track changes in annual productivity over a 5-year period in natural marshes in South Carolina.

When choosing a surrogate measure, it is critical to test the correlation with biomass initially. Such correlations may be relatively poor in diverse communities composed of species with multiple growth forms. For example, in a greenhouse microcosm experiment with randomly chosen 1-, 3-, and 6-species assemblages from southern California salt marshes, height, cover, and layering density were all poor predictors of standing crop (PERL, unpublished data).

6.6.14 Belowground biomass

Belowground biomass can be estimated by collecting marsh soil samples with large cores (Section 6.4.2). Choice of core size will depend upon goals and site characteristics. Core diameter should be sufficient to capture heterogeneous root distributions, with size balanced against the difficulty of extracting larger cores and the relative disturbance of core collection. Ten-cm diameter cores have been used in multi-species experiments at the Tidal Linkage at Tijuana Estuary, as well as in many other estimates of belowground biomass (de la Cruz and Hackney 1977, Roman and Daiber 1984, Hackney and de la Cruz 1986). Larger cores are more difficult to drive into the soil and extract, although 15-cm and larger cores have also been used in sampling belowground biomass (Schubauer and Hopkinson 1984). Samples are typically sectioned into 1- to 10-cm intervals to evaluate the relative distribution of belowground biomass by depth. To estimate productivity from biomass, samples must be sorted into live and dead material, and this can be very time consuming. Hsieh and Yang (1992) outline a method for distinguishing live and dead roots based on the release of dissolved organic carbon during boiling. This method correlated well with data from manual sorting and was much less labor intensive. Neill (1992) used mesh bags to measure root ingrowth in a prairie wetland, and this method also could be useful in coastal wetlands.

We suggest taking cores shortly after tidal inundation while the site is near saturation to ease core collection. Use of a vent to equalize pressure at the bottom of the core will make core removal much easier (Gallagher 1974). Core depth will depend upon root distribution, with the roots of most marsh plain species located within 20 cm of the surface, and few roots extending below 30 cm (Purer 1942, Keer 1999). In the higher marsh areas, roots may grow deeper to access water, so that deeper cores may be needed. Following collection, cores should be processed immediately, or they must be refrigerated to minimize decomposition until they are processed. Roots are separated from the soil and biomass relativized on an aerial basis (g/m^2).

Root washing methods should be tailored to individual needs and resources. We use a four-step process:

- break up the soil core with water spray in a 15-liter tub, and float the roots in the resulting suspension of clay and fine silts;
- pour off the water and root suspension (leaving the sand to be discarded) through a box-framed stainless steel fine-mesh screen (0.36 to 0.45-mm opening) that will pass clay and fine silts while retaining roots;
- spray roots off of the screen and back into the tub for additional rinsing; and
- pour off the cleaned roots over the screen, to be bagged, dried, and weighed.

Root washing is time intensive; samples with high sand and/or root biomass have to be teased apart and refloated several times to separate sand grains from the fine roots, and samples with high clay content have to be poured off several times before the clays are separated from root biomass.

6.6.15 Algal abundance and productivity

Algal mats beneath the marsh canopy can be highly productive (Zedler 1980). The algal community may be an important component of the trophic structure because of its high productivity and digestibility (Zedler 1980, 1982*a*). Algal abundance and growth also are useful indicators of eutrophication and tidal flushing. Although there is no easy way to obtain accurate estimates of algal abundance, algal dominants should be identified by type (e.g., drift macroalgae *Enteromorpha*, *Ulva*, green epibenthic mats of *Enteromorpha*, cyanobacterial epibenthic mats, and diatom films). Cover and standing crop estimates should also be made to convert short-term rates into estimates of long-term productivity under similar conditions. This allows comparison among or within algal communities through time. Cover or biomass of floating algal mats may be the most useful attributes, if the issue is eutrophication and differences are large between comparison sites.

Macroalgal mats can develop in tidal creeks, on mudflats, and drifting or growing within the emergent marsh (Fong 1986, Rudnicki 1986), while thick mats of filamentous blue-green algae and diatoms grow extensively on the intertidal mudflats (Zedler 1980, 1982*b*). In tidal channels, the highest algal biomass is measurable at low tide at the end of a neap tide series, when channels are less filled with seawater. More detailed studies of algal composition are needed to document biodiversity in the phytoplankton, edaphic algal mats, and floating algal mats (the larger green macrophytes support many species of epiphytes, as yet undocumented). About 100 species comprise the edaphic algal mats of Tijuana Estuary (Zedler 1982*b*).

Phytoplankton biomass is estimated as chlorophyll *a* concentration using either the fluorometric technique or extraction and measurement of absorbency using a spectrophotometer. Visual estimates of the percent of the water surface covered by macroalgae should be made at the permanent stations within the wetland, and the taxa noted (usually members of the Chlorophyta, either *Enteromorpha* or *Ulva*).

Algal productivity rates can be measured as algal O_2 evolution in chambers with a known initial dissolved O_2 concentration. Comparing changes in O_2 concentration after an incubation period in light and dark chambers yields an estimate of NPP. Newer methods of measuring productivity with microgradients of oxygen using oxygen electrodes should be evaluated before establishing a monitoring program for algal productivity. We have not found measures of chlorophyll in marsh soils to be predictive of any environmental conditions, perhaps because soil algae can grow in favorable conditions and persist as dormant cells for years afterward (Ross 1994). To project net productivity rates over a growing season, sampling must be relatively frequent to encompass variation due to seasonal and/or tidal dynamics (Zedler 1980).

6.6.16 Plant tissue nitrogen concentrations

The accumulation of nitrogen in newly established tidal wetlands is intimately tied to ecosystem development. Studies of constructed wetlands in San Diego County indicate that nitrogen is a key limiting nutrient to marsh plant growth (Langis et al. 1991, Boyer and Zedler 1998). Because the pool of nitrogen incorporated into organic material is increased as biomass and ecosystem complexity increase (Bazzaz 1975, Vitousek and White 1981), nitrogen pools indicate how marshes are developing. Boyer and Zedler (1998) found that the standing crop of nitrogen (concentration × biomass) was consistently higher in natural cordgrass stands than in a constructed marsh. Even where fertilizer addition increased nitrogen standing crop to levels in reference marshes, cordgrass returned to pre-fertilization conditions 1 year after fertilization ceased. The coarse mineral soils found in many constructed marshes may not supply or retain sufficient nitrogen to maintain cordgrass growth (Langis et al. 1991). As a result, the functions provided by tall, nitrogen-rich cordgrass stands might not develop within the typical 5-year monitoring period (Boyer and Zedler 1998, Zedler and Callaway 1999). Soil nitrogen sampling is discussed above (Section 6.4.10); here we outline methods for measuring tissue nitrogen pools.

Root samples are collected as described above (Section 6.6.14). Large spatial heterogeneity in belowground tissue nitrogen should be expected, once the restoration site develops vegetation. It is thus desirable to sample widely for representative accumulation patterns. Root tissue concentrations are determined from the roots washed from cores. For aboveground samples, representative tissue samples are collected from multiple, randomly chosen individuals of each species. Depending on plant morphology and the level of detail desired, tissues (roots, leaves, stems, etc.) can be composited or analyzed separately. Dried tissue samples are homogenized (e.g., with a Wiley Mill™) to pass through a 40-mesh screen. Minimum tissue mass for roots or shoots should be 0.20 g dry weight, although smaller samples can be used if they are nitrogen rich. Tissue concentrations for samples less than 0.10 g dry weight might not be reliable. Homogenized samples are digested following standard Kjeldahl digestion procedures (Allen 1989) and the supernatant is analyzed for total Kjeldahl nitrogen (TKN). Total nitrogen concentration can also be measured directly with a CHN analyzer. TKN represents all of the NH_3-N in a tissue sample, including organically bound forms plus inorganic NH_4^+ (NO_3^-, NO_2^- are not included).

6.7 Invertebrates

Aquatic invertebrates provide food web support for fishes and shorebirds. Because of their importance in food webs, invertebrate populations are frequently assessed as an indicator of restoration progress. Invertebrates also affect benthic processes such as erosion, sedimentation, and nutrient cycling. The benefits of these functions are less certain and not often quantified. As we learn more about how invertebrates affect the development of restoration sites, the need to assess their presence and abundance will become clearer.

6.7.1 Reference site selection

Reference sites are generally selected and sampled simultaneously with restored or created marshes. Reference sites should be selected with attention to the factors that influence invertebrate species composition and density. Sediment type has a strong influence on invertebrate distributions (Sanders 1958, Craig and Jones 1966); thus, sediment grain size

should be similar for reference and restored areas. Water flow influences sediment texture and the availability of suspended foods. Water depth influences the distribution of some nektonic invertebrates, in part by affecting their predators (Ruiz et al. 1993). The presence or type of submerged aquatic vegetation is also important. If a good match cannot be made between reference and restored site characteristics, then we recommend using a range of reference sites, from which a range in levels of performance can be derived.

6.7.2 Sample timing

Invertebrate assemblages vary in composition and abundance over multiple temporal scales, so the timing of sampling can significantly influence results. Temporal scales include diurnal variation due to water column migration or other activity patterns, seasonal variation due to reproductive timing, and interannual variation due to the effects of extreme climatic events. In southern California lagoons and estuaries, many species have reduced abundance (or temporary disappearance) during winters with unusually heavy rainfall; others show major, but brief, peaks in abundance following such events (Onuf 1987, PERL unpublished data). ENSO can also bring unusual species to estuarine habitats (Talley *in press*).

The monitoring program needs to include periods with and without unusual conditions. If frequent sampling is not possible, then the timing should be guided by knowledge of invertebrate life histories. For instance, to quantify juvenile stages, sampling at the peak of spawning may be the best approach. Data from elsewhere in the region should be useful in planning sampling to coincide with peaks in abundance or species richness.

6.7.3 Sampling methods

The approach and gear used to sample invertebrates depend on the mobility of the organisms being studied, their size, and the habitat they occupy. Here we discuss the methods used to assess some of the major groups of invertebrates found in wetland and estuarine habitats. Because mosquitoes are rarely encountered in our southern California tidal wetlands, we do not present methods for this taxon.

6.7.3.1 Infauna

Infauna can be collected from sediments using any number of devices (see Holme and McIntyre 1984), depending on the depth of the water over the sampling site. Our focus is on methods appropriate for shallow-water habitats, which generally require some type of coring device (Figure 6.10).

The depth to which samples are collected depends primarily on the biology of the organisms in question. In most cases, the majority of infauna are located within the upper 2 to 5 cm of the sediment surface (LaSalle et al. 1991, Levin et al. 1998). Some larger organisms can burrow to 60 cm (Peterson 1977); examples are burrowing shrimp (e.g., *Neotrypaea californiensis*) and bivalves (e.g., *Tagelus californianus*).

The techniques used to separate invertebrates from the sediments are as important as the depth of the sample because mesh size and sieving techniques can substantially influence density estimates (Reish 1959). The appropriate mesh size depends on the size of sediments to be rinsed and the biological questions being asked. Although some studies have used or recommended a 0.25-mm (Sacco et al. 1994), 0.3-mm (Moy and Levin 1991, Levin et al. 1996, 1998) or 1-mm mesh screen (Cammen 1976, PERL 1990, Simenstad et al. 1991), many studies of constructed salt marshes have used a 0.5-mm screen (LaSalle et al.

Figure 6.10 Replicate cores are taken to sample benthic invertebrates at Tijuana Estuary.

1991, Minello and Zimmerman 1992, Havens et al. 1995, Simenstad and Thom 1996, Minello and Webb 1997, Posey et al. 1997).

Efforts should be made to match sampling methods with those used for collecting long-term reference data. If reference data were collected using an inappropriate mesh size for the questions at hand, then using nested screens of different sizes is an alternative (Levin et al. 1998). Sampling with two types of cores (5- and 20-cm depth) and two mesh sizes (0.5- or 1.0- and 3.0-mm, respectively) allows for estimates of both shallow and deep-dwelling organisms without a significant increase in sorting time (PERL 1990). The 20-cm "deep" cores can be sorted in the field, with bivalves and burrowing shrimp counted and released. Bivalve sizes can also be measured to provide additional useful information on population structure and recruitment.

Replicate cores should be taken within sampling stations because infaunal populations are usually patchy (Hewitt et al. 1993, Thrush et al. 1994). A pilot study to determine the number of replicate cores necessary for an accurate density estimate is advisable before beginning monitoring (Thrush et al. 1994, Streever 1998). At Tijuana Estuary, five cores per station were considered adequate (PERL 1990), but more cores may be required in other areas. Accurate density estimates may require ten or more cores per station (Thrush et al. 1994). The number of replicates required will vary depending on the heterogeneity of the sampling station and other factors.

6.7.3.2 Epifauna

Some epifaunal taxa, such as crabs and snails, are either too large or too mobile to be sampled with sediment cores. Others are associated with marsh vegetation, rather than sediments. Various methods have been used to sample these mobile epibenthic taxa. For

sessile animals that cover benthic surfaces (e.g., *Ostrea* spp.), percent cover can be monitored (Simenstad et al. 1991). The California horn snail (*Cerithidea californica*) has been counted in quadrats placed on the sediment surface (Race 1981, Levin et al. 1996); the quadrat method has also been used to quantify the burrows of the fiddler crab, *Uca crenulata*. Other crabs, such as yellow shore crab *Hemigrapsus oregonensis*, have been successfully sampled from baited pit traps. Litterbags make good traps for the smaller epifaunal organisms of the vegetated marsh (Scatolini and Zedler 1996). Litterbag traps consist of mesh bags filled with dried plant material (e.g., *Spartina foliosa*), which are deployed on the marsh surface over a period of weeks to months. Litterbags become colonized by snails and small crabs, as well as many invertebrates associated with vegetation (e.g., isopods, insects, and amphipods). Although pit traps and litterbags do not provide quantitative estimates of abundance, they are useful in comparing relative abundance between habitat types, such as between natural and constructed marshes (Scatolini and Zedler 1996).

6.7.3.3 Nektonic invertebrates

Actively swimming nektonic invertebrates, such as shrimp and crabs, include some of the most commercially important species found in estuarine and salt marsh habitats. These taxa can evade sampling gear used to collect sedentary or slow-moving invertebrates. Nektonic invertebrates may be collected using beach seines or trawls, or surveyed using snorkel or SCUBA transects (Simenstad et al. 1991). The type of gear selected will depend on the type of substrate present and the tidal regime (intertidal or subtidal) of the habitat to be sampled. For beach seines and trawls, mesh size is an important consideration. For survey transects and trawls, the distance sampled should be consistent throughout the sampling program. Drop or throw samplers used in conjunction with dip nets are frequently employed in vegetated habitats (Minello and Webb 1997). This procedure generally requires clipping of vegetation, which may not be advisable in small or highly impacted marshes.

6.7.4 Sampling parameters

As with fish (Section 6.8), a number of invertebrate assemblage parameters can be measured to evaluate the functioning of restored coastal wetland sites. Most published studies have relied on some measure of density (either total density or density of dominant or commercially important taxa) as well as some measure of diversity, such as species richness or a diversity (i.e., dominance or evenness) index (Table 6.1). Fewer studies have used population attributes, such as size structure or biomass, or community attributes, such as trophic structure (Table 6.1). No studies have examined the growth or feeding of invertebrates in restored habitats. These parameters may provide important information on the functioning of restored habitats (Moy and Levin 1991, Peck et al. 1994). A more thorough examination of the population and community attributes of invertebrate assemblages in restored marshes would help define the functioning of these habitats.

6.7.5 Sample identification

Sorting and identification of invertebrates often take up large amounts of time, particularly for infaunal samples. In one case, these activities were estimated to cost 12× that of the initial sample collection (Saila et al. 1976). The taxonomic level chosen for identification of specimens (species, genus, family, or higher) has a substantial effect on the time required for identification, so this is an important decision in planning a monitoring program.

Table 6.1 Summary of studies from the recent literature which have evaluated infaunal, epifaunal, or nektonic invertebrate assemblages of created and restored salt marshes along the Atlantic, Gulf, and Pacific coasts of the U.S.

	Site Characteristics				Sampling Frequency		Taxa Sampled			Parameters Measured*							
State	Citation	Size (ha)	# Sites Sampled	Age (yrs)	# Samples per yr.	# Yrs	Inf	Epi	Nek	DT	DI	SR	D	B	SS	TS	DM
	Atlantic Coast																
CT	Peck et al. 1994	20	1	13	2	2		X			X			X	X		
VA	Havens et al. 1995	2.2	1	5	2	1	X	X	X	X	X	X	X		X		
NC	Cammen 1976	NS	2	1–2	4–5	1	X			X		X	X	X			
NC	Moy and Levin 1991	0.2	1	3	2	2	X			X	X					X	
NC	Sacco et al. 1994	0.1–0.5	6	1–17	1	1	X			X	X					X	
NC	Levin et al. 1996	0.9	1	5	1–5	5	X	X		X	X	X	X			X	X
SC	LaSalle et al. 1991	35	2	4–8	1	1	X		X	X	X	X	X				
SC	Posey et al. 1997	NS	3	7–15	1–2	2	X				X						
	Gulf Coast																
TX	Minello et al. 1994	8	1	5	1–2	2	X	X	X	X	X	X					
TX	Minello and Zimmerman 1992	5–8	3	2–5	1	1	X	X	X	X	X		X				
TX	Minello and Webb 1997	0.1–10.5	10	3–15	2	1	X	X	X	X	X	X			X		
	Pacific Coast																
WA	Simenstad and Thom 1996	3.9	1	8	1–2	7	X	X		X		X		X			
CA	Scatolini and Zedler 1996	4.9	1	4	8	1		X		X		X					
CA	Zedler 1996b	4.9	1	10	1–4	6	X			X		X					

* NS = not specified; Inf = infauna; Epi = epifauna; Nek = nekton
DT = density — total; DI = density — individual taxa; SR = species richness; D = diversity
B = biomass; SS = size structure; TS = trophic structure; DM = developmental modes

Although organisms are often identified to the species level, this level of accuracy may be unwarranted (McIntyre et al. 1984, Ferraro and Cole 1990). In addition, data on rare species are generally not useful in multivariate analysis; if this type of analysis is to be employed, then it may be advisable to enumerate only common species or taxa. Rare species can be preserved for later identification.

6.8 Fishes

Fish are highly mobile animals that use a number of wetland habitats over a variety of time scales for refuge, feeding, reproduction, or other functions. Assessing the magnitude and duration of their requirements in both natural and restored habitats is a difficult endeavor, and it is generally accomplished by measuring corresponding structural attributes of the fish assemblage. In general, the more rigorous measures of fish habitat functionality require the most time and financial resources (Table 6.2). The least difficult and least costly sampling methods merely tally species occurrences in order to derive some measure of habitat suitability and refuge functions, while data on fish diets and growth rates are needed to estimate the habitat's ability to provide adequate food-chain support and contribute to individual fitness.

Predictably, most wetland restoration projects assess fish habitat function by measuring fish occurrence or density, while few quantify diet or growth rates. Since 1990, 15 peer-reviewed papers examined restoration of fish habitat in tidal marsh systems, of which 10 relied upon quantification of total fish density, while only 2 actually estimated fish growth (Table 6.3). These published studies represent the most rigorously monitored wetland habitat restoration/mitigation projects conducted (out of thousands) throughout the U.S.

In practice, fish support functions are evaluated in only a tiny proportion of restoration sites. Fish rapidly colonize new sites, often leading to the perception that such sites are fully restored. However, species occurrence or abundance may not accurately indicate habitat value (Minello and Webb 1997). Projects that rely solely on fish presence may provide misleading information on restoration progress (Zedler and Callaway *in press*). General measures also mask important aspects of individual species health, assemblage structure, and community ecology. In contrast, growth rate is a generally accepted measure of fitness and a quantitative response to a variety of integrated environmental factors. Fish habitat assessment procedures can be improved by requiring these better estimates of habitat functioning.

Table 6.2 Structural attributes and associated fish habitat functions commonly used in assessing a site's functional equivalence, broadly ranked by collection effort difficulty and cost (1 = low, 6 = high).

Rank	Structural Attribute	Function
1.	Species occurrence (qualitative)	habitat suitability, exotics
2.	Density; standing stock, population growth rate (quantitative)	habitat quality and refuge
3.	Population structure	reproduction, recruitment, nursery role
4.	Residence time	movement, emigration and immigration
5.	Diet	feeding, feeding success, food chain support
6.	Individual growth rate	fitness

Table 6.3 Summary of studies from the recent literature which have quantified fish parameters to indicate the function of restored and created wetland habitats.

		Site Characteristics					Sampling Frequency		Parameters Measured						
State	Citation	Control (Y/N)	Size (ha)	# Sites Sampled	Type of Rest.	Age (yrs)	# Samples per yr.	# Yrs	SR	DT	DI	PS	RT	D/F	G
					Atlantic Coast										
NC	Moy and Levin 1991	Y	0.24	1	excavate	3	5	3						X	
NC	Rulifson 1991	Y		2	excavate	5	monthly	5	X	X	X				
SC	LaSalle et al. 1991		35	2	dredge	4 and 8	1	1	X	X	X			X	
VA	Havens et al. 1995	Y	2.2	1	excavate	5	2–3	1	X	X	X	X			
					Gulf Coast										
TX	Minello and Zimmerman 1992	Y	5–8.1	3	dredge	2–5	1	1	X	X	X				
TX	Minello et al. 1994	N	8	1	dredge	5	3	2	X	X	X				
FL	Vose and Bell 1994	Y	1.6	1	dike	0–2	monthly	2	X	X	X				
TX	Minello and Webb 1997	Y	0.1–10.5	10	exc/dredge	3–15	2	1	X	X	X				
FL	Llanso et al. 1998	Y	1.6	1	dike	0–3	variable	3						X	
					Pacific Coast										
WA	Shreffler et al. 1990	N	3.9	1	excavate	3	variable	2					X		
WA	Shreffler et al. 1992	N	3.9	1	excavate	3	variable	2						X	X
CA	Chamberlain and Barnhart 1993	Y	3.5	1	dike	1–2	monthly	1.3	X	X	X	X			
WA	Simenstad and Thom 1996	N	3.9	1	excavate	8	variable	7	X	X	X				
CA	Williams and Zedler 1999	Y	4.9–7	4	excavate	6–12	annually	8	X	X	X				
WA	Miller and Simenstad 1997	Y	1.6	1	excavate	1	variable	2					X	X	X

* SR = species richness DT = density — total DI = density — individual taxa PS = population structure
RT = residence Time D/F = diet/feeding G = growth

Box 6.3 Estuary management and habitat value for juvenile California halibut: predictions of growth using a bioenergetics model

Sharook Madon

In southern California, estuaries, bays, and lagoons provide critical habitat and nursery functions for juvenile California halibut, *Paralichthys californicus*, a highly prized commercial and recreational species (Kramer 1991) and valued indicator organism (Emmett et al. 1991). Yet, we know little of how environmental variations within these habitats control halibut growth. Large variations in rainfall, streamflow, salinity, and water temperature (e.g., via inlet closures and runoff) potentially affect halibut growth directly, by influencing energy intake and metabolism, and indirectly, by altering or reducing prey resources. Information on how such variations control halibut growth is essential for guiding the restoration of tidal flushing to coastal water bodies that sometimes close to tidal action (e.g., Los Peñasquitos Lagoon, Santa Margarita Estuary), as well as nontidal areas being restored to tidal action (e.g., Bolsa Chica Wetland in Long Beach, California). The high cost of some proposed restoration efforts ($60 million for Bolsa Chica Wetland) makes the prediction of increased habitat value rather urgent.

Our earlier experiments of the effects of decreased salinity on juvenile California halibut showed that metabolic rates of halibut increase two-fold when salinity decreases from 34 to 17 ppt and four-fold when salinity drops from 34 to 8 ppt (Baczkowski 1992). Such salinity-induced stresses represent substantial energy costs, likely affecting halibut growth where salinity drops precipitously. Such sudden changes are known to occur when a river mouth closes to the ocean and freshwater continues to flow into the coastal water body (Nordby and Zedler 1991). How salinity affects halibut growth is also determined by fish size, water temperature, and food intake rates, which are dependent on prey availability. The interactive effects of these variables on halibut growth are often difficult to visualize without use of bioenergetics models.

We used a bioenergetics model for *P. californicus* (see Table 6.4 for model parameters) to evaluate the combined effects of salinity, water temperature, and prey availability on the growth potential (g/g/d) of small (1 g mass, 40 mm TL) and large (80 g mass, 200 mm TL) juvenile halibut. The model simulated the effects of a 5-day mouth closure event at Los Peñasquitos Lagoon, based on known effects of salinity reductions on halibut metabolic rates (Baczkowski 1992). Our model simulations indicate that larger halibut are more severely affected by variations in salinity and water temperature than are smaller halibut. When prey availability is low and water temperatures are between 15 to 25°C, a drop in salinity below 25 to 27 ppt causes halibut to lose weight. Higher levels of prey availability allow halibut to compensate for salinity-induced increases in metabolic costs. Small halibut can tolerate a greater degree of salinity depression than large halibut; however, at temperatures exceeding 25 to 26°C, both large and small halibut lose weight, even when salinity is near seawater concentration.

Los Peñasquitos Lagoon (LPL, Box 1.4) is a small coastal wetland that functions as nursery habitat for California halibut. Intermittently, it closes to tidal flushing. In recent years, increased flows of freshwater into the lagoon during periods of mouth closure lead

Table 6.4 Parameters for the juvenile California halibut, *Paralichthys californicus*, bioenergetics model. Parameters were estimated from Ehrlich et al. (1979), Innis (1980), Drawbridge (1990), Baczkowski (1992), Caterino (1992), and ongoing laboratory studies (Madon et al., *unpublished data*).

Symbol	Parameter Description	Parameter Value
	Consumption	
a_c	Intercept for maximum consumption (g g^{-1} d^{-1}, wet wt.)	0.641
b_c	Exponent for maximum consumption	–0.468
Q_c	Slope for temperature-dependence of consumption	3.6
T_{copt}	Optimum temperature for consumption for larvae	25°C
T_{cmax}	Maximum temperature for consumption	28°C
	Respiration	
a_T	Intercept for routine respiration (g O_2 g^{-1} d^{-1})	0.009
b_r	Exponent for routine respiration	–0.304
Q_r	Slope for temperature-dependence for routine respiration	2.9
T_{ropt}	Optimum temperature for routine respiration	27°C
T_{rmax}	Maximum temperature for routine respiration	30°C
S	Specific dynamic action coefficient	0.172
A	Activity parameter	1.0
	Egestion and excretion	
a_f	Proportion of consumed food egested	0.15
a_u	Proportion of assimilated food excreted	0.07
	Caloric Densities (calories g^{-1} wet wt.)	
	Halibut	1200
	Benthic invertebrate prey	1000
	Goby prey	1100

to hyposaline conditions. A 5-day mouth closure occurred in May 1997. Beforehand, LPL was open to tidal flushing, and average daily water temperature and salinity ranged from 22 to 23°C and 30 to 31 ppt, respectively. When the mouth closed, daytime water temperature increased to 25 to 26°C, and salinity declined to 26 to 27 ppt for the 5-day period. The model predicts that, when prey availability is low (as is likely when tidal flushing ceases), growth of small halibut is reduced by 10% while growth of larger fish is reduced by 51% over this 5-day period. In contrast, when prey availability is high (less likely), growth of small halibut is still reduced by 8% and that of large fish declines by 16%.

Our halibut model illustrates the complex manner in which water temperature, salinity, prey availability, and fish size affect halibut growth. By simultaneously considering all these factors, predictions from the model could be used to set targets for restoring and maintaining coastal wetlands as viable nursery habitats. Managing the inlet mouth to minimize closure time should also enhance survival of stenohaline invertebrate prey species (Nordby and Zedler 1991), ensuring adequate food supply for small juvenile halibut. For example, restrictions on mouth-closure times could be set to minimize salinity- and temperature-related stress on this species. Restrictions are especially needed during the summer, when mouth closure leads to warm, hypersaline water. In winter, closure is also a problem if rainfall occurs during periods of reduced tidal flushing; this combination creates hyposaline conditions, which are also stressful to many species.

Tidal closure can be corrected, at least for brief periods, by calling in bulldozers to remove accumulated sand at the inlet. It is preferable, of course, to prevent closure by providing a large tidal prism. For LPL and many other small lagoons, this is no longer

an affordable option. However, it should be possible to incorporate well-flushed, deep-water channels into restoration sites so that sensitive species have a refuge from elevated summer temperatures (>25°C) and variable salinity.

6.8.1 Reference site selection

Developing a restoration model (Chapter 2) that provides full fish support functions requires knowledge of fish habitat requirements and of natural marsh habitat ratios. Factors such as hydrology and habitat structure (channel morphology) are known to influence the fish assemblage in marsh habitats (Rozas and Odum 1987, McIvor and Odum 1988, Baltz et al. 1993, Williams and Zedler 1999; Chapter 5). It is of paramount importance that reference sites be selected with similar habitat and landscape features (Figure 5.3). While marsh habitats comprise a continuum, some level of habitat categorization is needed to describe fish functions and measure associated attributes. Categorization is also important because naturally heterogeneous marsh habitats and the diverse behaviors of different species may preclude use of one method or more to sample fish populations.

6.8.2 Sample timing

The frequency and duration of sampling should relate to monitoring goals and a statistically valid sampling design. If funds limit sampling to once per year, it should coincide with annual peaks in abundance (i.e., the summer in most regions of the U.S.). Conversely, if information on recruitment and growth is desired, it is necessary to sample multiple times per year. If historic reference data are available for the sites of interest, efforts should be made to standardize sampling methods and schedules (i.e., tidal stage, mesh size, seasonal conditions, etc.).

Long-term monitoring data (>5 to 10 years) are highly desirable because population abundances of estuarine animals, especially those with long and complex life cycles, are extremely variable through time and space. Short-term (interannual) variability of estuaries is high, obscuring chronic and cumulative effects of human activities (Hines et al. 1987). Often, rare or infrequent events (e.g., ENSO events, floods, or hurricanes) occur at long, multiyear intervals that are not captured with short, annual studies.

6.8.3 Sampling gear

The selection of a sampling method must be appropriate for the target species and habitats, as well as for the objectives of the study. Summaries of common techniques used to assess fish population parameters in estuarine habitats follow. More extensive descriptions of sampling methods and gear characteristics are included in Simenstad et al. (1991), Murphy and Willis (1996), Kneib (1997), and Rozas and Minello (1997).

Deep, subtidal areas with unvegetated bottoms are generally sampled with otter- and beam-trawls, which are towed behind a boat and can sample a large area (Allen 1982, Gunderson and Ellis 1986). Towed nets produce quantitative estimates of fish populations with apparent ease, although catch efficiencies can be low and are influenced by many factors (e.g., species composition, environmental characteristics, substrate; Rozas and Minello 1997).

In tidal channels (0-2 m deep), fish assemblages have been quantitatively sampled with large beach seines (Figure 6.11; PERL 1990, Nordby and Zedler 1991, Williams and Zedler 1999). Repeated passes with a bag seine through a channel area enclosed by

Figure 6.11 A large beach seine is used to sample fishes quantitatively in a large channel at Tijuana Estuary.

blocking nets showed that the majority of fishes were removed with this method, although catchability varied by species (Nordby and Zedler 1991). Samples from entire channel cross-sections combine a number of discrete habitats, however, and may reduce the ability to identify distinct species-habitat associations.

For most shallow (<1.0 m deep) estuarine habitats (including small tidal creeks and the marsh surface), Rozas and Minello (1997) strongly recommended use of enclosure samplers (e.g., drop nets and throw traps) for estimating densities of small fishes and crustaceans (Wegener et al. 1973, Zimmerman and Minello 1984). These samplers provided the most accurate and efficient estimates of nekton density in discrete habitat units, and estimates were not significantly influenced by the presence of vegetation. However, they were effort-intensive for a very small sample area, and frequently involved removal of vegetation. Destructive sampling methods could potentially have a great impact on remaining populations of rare or endangered species in sensitive habitats (e.g., southern California's small and fragmented coastal wetlands) where passive sampling methods might be warranted.

Several passive sampling methods have been used to assess fish use of small tidal channels and the marsh surface. These include channel, flume, and fyke nets and pit traps, which collect nekton on falling tides (Figure 6.12; Cain and Dean 1976, Levy and Northcote 1982, Kneib 1984, Hettler 1989, Shreffler et al. 1990, Rountree and Able 1992, Desmond et al. 1999). While all of these methods sample large habitat areas fairly nondestructively, their catch efficiency is largely unknown and density estimates are not easy to determine. Further, they are highly selective in the species and size of animals trapped and likely depend on habitat characteristics. Flume nets, flume weirs, and lift nets are recent methodological improvements that quantitatively sample nekton on intertidal marsh surfaces (McIvor and Odum 1988, Kneib 1991, Rozas 1992).

Figure 6.12 Channel nets are set during high tide at the mouth of small intertidal creeks at Sweetwater Marsh. As the tide ebbs, fish are funneled into the net.

6.8.4 Population structure

Population structure can be an important indication of restored habitat function to fish populations, providing information on recruitment, juvenile nursery refuges, and reproduction (Simenstad et al. 1991). Life-history information, such as the timing of recruitment events, can be provided by length and weight measurements which differentiate individuals into distinct size classes that can be related to age (Balon 1975). Sex, sexual maturity, and spawning state can be made by internal examination of the gonads.

6.8.5 Residence time

For mobile fishes, the period of time an animal occupies a habitat may be an important measure of the habitat's functional support (Simenstad et al. 1991). In some species of juvenile salmon it has been argued that estuarine residence time is an important survival factor, and that time spent in specific habitats can be an indication of the habitat's relative value (Simenstad et al. 1982). Fish movements can provide essential information on habitat linkages in a heterogeneous landscape, thereby improving plans to restore, preserve, and/or manage large, complex ecosystems (Irlandi and Crawford 1997). Fish residence time and movements are most often determined by mark-recapture methods, which are more easily conducted in closed or semi-closed systems (see Weinstein and O'Neil 1986, Shreffler et al. 1990). Fish marking methods include fin clipping and insertion of coded wire tags (Nielson and Johnson 1983), subcutaneous injection of dyes (Lotrich and Meredith 1974, Thresher and Gronell 1978), spraying with fluorescent pigments (Bax 1983, Shreffler et al. 1990), and otolith marking (temperature-based, Volk et al. 1994; chemical-based, Monaghan 1993, Miller and Simenstad 1997).

More recently, techniques for determining stock origins, habitat utilization, and migration patterns have been developed via the use of otolith elemental fingerprinting (Edmonds et al. 1989, Campana et al. 1995, Secor et al. 1995, Thorrold et al. 1998). The future application of otolith microchemistry methods to estimate habitat utilization could provide important answers to a variety of ecological and management-oriented questions in restored habitats.

6.8.6 Diet and feeding

Diet and feeding studies are accomplished through stomach content analyses and the direct observation of fish feeding activities. Results can be used to describe and identify trophic linkages, energy flow, and organic matter sources that support consumers in a restored system. Feeding studies should quantitatively assess diets at the population level by documenting prey identity, number, and biomass (volume; Pinkas et al. 1971, Hyslop 1980). When used with local estimates of prey availability, feeding studies can be used to calculate prey selectivity (Vanderploeg and Scavia 1979, Lechowicz 1982, Llanso et al. 1998) and assess preferred prey types. Stable isotope signatures (C, N, S) of predators, prey, and primary producers can also be used to further trace energy sources, organic matter flow, and trophic relationships (e.g., Peterson and Fry 1987, Kwak and Zedler 1997, Page 1997).

6.8.7 Growth rate

As a quantitative response to a variety of integrated environmental factors (e.g., prey availability and value, predation risk, physicochemical parameters), fish growth is an excellent parameter for assessing the function of restored habitats. Mean growth rates (length, weight) can be derived for a population of fish by using mark-recapture techniques (Sogard 1992). Use of fish otolith microstructure has also been used to compare the relative growth of fishes in a restored habitat compared to a natural habitat (Miller and Simenstad 1997). Other analytical techniques that estimate growth (e.g., RNA-DNA ratios, Buckley 1984) exist but have yet to be applied in a restoration assessment context.

6.9 Recommendations for minimum monitoring

We recognize that most assessment and monitoring programs will be constrained by funding and by the availability of personnel who are qualified to sample complex processes, such as nitrogen fixation. We are frequently asked to recommend minimal requirements for monitoring sites to see if they have met permit conditions for wetland restoration or construction. Thus, we have prioritized items based on what one most needs to know and how much information is provided by the data (Table 6.5, where priority 1 = most needed; 2 = desirable; 3 = worthwhile). Monitoring programs can be expanded or reduced by varying the number of attributes examined, the frequency of examination, and the number of sampling stations (Table 6.6). Additional variables include the detail of examination within sampling stations (e.g., depths at which soil salinity is measured) and at the laboratory (e.g., determination of invertebrates to family or to species; chemical analysis of pooled or individual soil samples from each sampling station).

In general, it is best to begin with an expanded monitoring program and then use the data to determine where effort can be reduced. Reduction might be brought about by sampling fewer attributes, using fewer stations (Section 6.9.1), or sampling sites less frequently. No two wetlands will have the same spatial and temporal variability, so no single program can fit all systems. Soil salinity, for example, will show greater extremes and more sudden changes in lagoon wetlands (often nontidal) than in fully tidal marshes.

Table 6.5 Priorities for wetland attributes to be monitored. Priority 1 = most needed.

Attribute and measures	Priority	Frequency of measurement; notes
Hydrology and Topography		
Inundation regime	2	spring tide cycle, e.g., in Nov.
Ground water levels	3	seasonal; as research study
Flow rates	3	spring tide cycle, e.g., in Nov.
Elevation	1	initially; thereafter coordinated with sedimentation monitoring
Sediment accretion and erosion	1	annually or after storms or floods
Creek morphology	2	initially; annually thereafter at permanent cross sections
Water Quality		
Water temperature and dissolved oxygen	2	seasonal or following specific events, such as tidal inlet closure; with datalogger over two-week tide series where possible
Water salinity	1	monthly at same tide condition; surface and bottom; with datalogger where possible
Water pH	2	sampled with water salinity
Turbidity and water column stratification	3	seasonal; as research study
Nutrients	2	seasonal
Soils		
Water content	2	seasonal; potential indicator of plant stress in high marsh
Bulk density	2	initially; following sedimentation events
Soil texture	1	initially; following sedimentation events
Soil salinity	1	seasonal (at least April and September)
Soil pH	1	initially; to detect soil acidification
Redox potential	2	useful to diagnose cause of plant stress
Organic matter	1	preplanting to determine need for amendments; annually thereafter
Total Kjeldahl nitrogen	1	preplanting; annually thereafter
Inorganic nitrogen	3	seasonal; as research study
Nitrogen processes	3	seasonal; as research study
Phosphorus	2	seasonal; where P may be limiting
Decomposition	3	seasonal; as research study
Toxic substances	3	pre-construction; cost permitting
Vegetation		
Vegetation mapping	1	annually; with aerial photos or remote data
Cover and height of vascular plants	1	annually, at the time of peak biomass
Canopy architecture		
Cordgrass height and total stem length	1	annually, at the time of peak biomass
Layering of vascular plants	2	annually, at the time of peak biomass
Patch size, distribution of target plants	1	annually, timing is species-specific; winter annuals in spring
Aboveground biomass	3	annually; not recommended except with non-destructive sampling methods
Belowground biomass	3	annually; as research study
Visual estimate of algal cover by dominant type	1	monthly; with salinity samples
Tissue nitrogen concentrations	3	annually, end of growing season; cost permitting

Table 6.5 (continued) Priorities for wetland attributes to be monitored. Priority 1 = most needed.

Attribute and measures	Priority	Frequency of measurement; notes
	Fauna	
Invertebrates		
Macrofauna (colonization rate, spp. composition, density)	1	seasonal (annually, at minimum)
Meiofauna (spp. composition, density)	2	seasonal, as research study
Insects (pollinators, predators)	3	seasonal; important where annual plants are required; census in spring or warm season
Fishes (colonization rate, spp. composition, density, size structure, growth)	1	seasonal if possible (June/Sept. at minimum); see Table 6.2 for other attributes to monitor
Birds		
Migratory periods	1	weekly in fall-spring; biweekly in summer
Nesting; fledging of young	2	during nesting period
Reptiles/Mammals	3	summer (high priority if rare spp. are present)

Monitoring programs should be tailored to the needs of the system being monitored, beginning with frequent measurements and reducing sampling when experience suggests that reducing the frequency will not significantly reduce information that is needed. Monitors should be prepared to increase sampling frequency in response to events such as floods, wastewater spills, algal blooms, or inlet closure. For determining if very specific mitigation criteria are being met, a highly tailored monitoring protocol would be needed, but the items of general interest would still be useful for understanding how the system functions.

6.9.1 Numbers of sampling stations

Field monitoring programs should provide an adequate sample of the area to which results will be generalized. Experienced field ecologists can usually walk through a site and delimit habitat areas that are "relatively homogeneous," but aerial photos are a great aid. Within each habitat area, replicate samples are needed at no fewer than three stations. Initial sampling will provide estimates of heterogeneity (variance of each attribute measured); if initial replicate stations give high variance (e.g., if the standard error exceeds 10% of the mean), additional replicate samples are needed to characterize the attribute adequately.

An alternative approach to replicate sampling within habitat areas is appropriate where gradients of environmental conditions are present. For estuarine channels that range from high salinity at the inlet to low salinity inland, it is more useful to position sampling stations along the gradient and to plot water quality characteristics against distance. Instead of clumping sampling stations within homogeneous sampling areas, one would distribute the stations at intervals proceeding upstream from the ocean inlet. Stations should be closer together where environmental changes are likely to be greatest. Results can be summarized as graphs of each attribute against distance from inlet, looking for spatial trends and evidence of shifts through time.

Table 6.6 A minimum monitoring program for a wetland with aquatic and marsh habitats.

	Before project	Annual monitoring schedule*											
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
		Hydrology and topography											
Elevation	**X**	and after sedimentation or erosion events											
Sediment accretion and erosion		annually or after storms or floods											
		Water quality											
Water salinity		**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**
		Soils											
Soil texture	**X**	and after sedimentation events											
Soil salinity		**M**			**M**			**M**		**M**			
pH		**X**	**I**										
Organic matter	**X**									**M**			
Soil TKN	**X**									**M**			
		Vegetation											
Vegetation mapping	**X**									**M**			
Cordgrass height and total stem length									**M**				
Cover and height of other vascular plants									**M**				
Patch size of target plants					**M**					**M**			
Algal cover by dominant type		**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**
		Fauna											
Fishes				**m**			**M**			**M**			**m**
Benthic invertebrates				**m**			**M**			**m**			**m**
Birds	**W**	**W**	**W**	**W**	**W**	**B**	**B**	**B**	**B**	**W**	**W**	**W**	**W**

* **X =** pre-project sample; **I =** initial sample only
M = sample once during month; **m** = sample to omit if funds are insufficient
W = sample weekly through month; **B** = sample biweekly

6.9.2 Sampling period

From the standpoint of the biota, a 20-year monitoring period is not unreasonable. It may take even longer for the restored marsh to develop its full potential as habitat for rare species, such as endangered birds. It may take longer for the soil organic matter to increase to natural levels. It may take longer for herbivory problems to become controlled by native predators. Finally, for a region that has highly variable rainfall, it may take 20 years to characterize the mode, or most usual condition of the wetland, should the monitoring period include years of unusual events. Such is the case at Tijuana Estuary, where salt marsh monitoring began in 1979 after nearly 40 years without flooding. Two major floods occurred in winter 1980, prolonged flooding occurred through April 1983, the tidal inlet closed for 8 months in 1984, and raw sewage inflows became substantial and continuous in about 1986. Flooding occurred repeatedly in the 1990s, and interannual variability has remained high.

Choosing a sampling program that can provide ecologically meaningful data through a 10 to-20-year period is not easy. Dozens of decisions need to be made, and most require careful judgment based on preliminary data from the system in question. The assistance of experienced field workers will be needed to tailor any "generic monitoring program" to the system in question.

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chapter seven

Sustaining restored wetlands: identifying and solving management problems

John C. Callaway and Gary Sullivan

7.1 Introduction

It is clear that fully functional wetlands are not easily created and that even functionally restored wetlands may not be self-sustaining (Zedler and Weller 1989, Mitsch and Wilson 1996). The more degraded sites tend to present the most difficult challenges and the least predictable outcomes (Chapter 1). Short-term expectations for progress of created and restored sites are often too high and unrealistic. As restoration proceeds and the site begins to develop, many maintenance issues are likely to arise, both foreseen and unforeseen. To learn from these developments and to move the science of restoration ecology forward, we recommend an adaptive approach (Chapter 1). Restoration can be made adaptive by:

- establishing realistic expectations,
- incorporating experiments into the project to understand better the cause/effect relationships that drive ecosystem development,
- assessing the site regularly,
- identifying shortcomings and their causes,
- making adjustments (mid-course corrections), and
- continuing to assess and adjust.

Below we discuss various maintenance problems that may occur as restored wetlands develop. If anything is certain in the restoration process, it is that unexpected problems will arise. A drought may impair high-marsh establishment, herbivores may damage newly planted vegetation, algal blooms may suffocate young seedlings, exotics may invade, storm events may cover the site in sediments, and human activities may intrude on the system. Such problems can threaten the progress of any project if they are not dealt with appropriately. Maintenance problems and their implications can be complex, and the consideration of multiple pathways of development should be incorporated into the planning process. For example, during the early stages of restoration, vegetation can be irrigated until the plants are well established, but lowered salinities and enhanced soil moisture may encourage exotic plant invasions. Areas disturbed by animals or people can

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be repaired and replanted, but the seasonal window of opportunity for planting may be narrow.

Because of the applied nature of restoration work, information on problem identification and mid-course correction has not been widely published. The maintenance of restored wetlands remains an area in need of further study and experimentation. Detailed evaluation of problems and the publication of findings will greatly enhance future restoration efforts.

7.2 Irrigation

Many wetland restoration projects regularly call for irrigation as part of the establishment phase, particularly in areas that are not regularly inundated by tides. In mediterranean climates, with low summer rainfall and high evaporation rates, irrigation is essential to protect new plantings from water and salt stress. As soils dry, porewater becomes more tightly bound to soil particles and increasingly unavailable to plants. Also, soil drying concentrates salts, compounding water stress by decreasing the potential for water uptake. Eventually plants cannot replace water lost through transpiration, and they may die. Even with regular irrigation, a single day of hot dry wind between waterings can overstress young plantings. The coarse soils that are typical of restoration sites facilitate desiccation and prolong the period of moisture stress (Section 4.3.3).

7.2.1 When to irrigate

Irrigation is most important directly following planting, and an irrigation system should be in place either before planting or immediately after it is completed (Figure 7.1). New plantings frequently have shallow, poorly developed, or pot-bound roots that are particularly susceptible to drought stress, especially during the interval between planting and

Figure 7.1 Irrigation setup at the Tidal Linkage, spring 1997.

the initiation of tidal action. Even within fully tidal sites, high-marsh plants are rarely inundated, and marsh-plain plants receive little inundation during seasonal cycles of low-amplitude tides (March and September in southern California). Hot, dry weather, coupled with minimal inundation, can parch the soil surface quickly, especially if it is sandy. Under such conditions, even the healthiest plants will be lost without irrigation unless they have established a deep root system. Where soil moisture is high from groundwater seepage, rainfall, or tidal inundation, site conditions can still vary unpredictably throughout a growing season.

Thus, we recommend an irrigation system for both high-marsh and marsh-plain habitats. Spray irrigation is most often used because it covers more area than drip irrigation, but if particular plantings need additional water or are widely spaced, drip irrigation can be useful. Irrigation should be viewed as a stopgap measure to be terminated once it is no longer needed.

7.2.2 Irrigation schedule

Several factors should be considered in planning the irrigation schedule for a particular site, including topography, soil texture, wind exposure, elevation, hydrology, and the type of vegetation present. The soaking period (duration of each watering event) should be sufficient to hydrate the soil to levels consistent with tidal action. Wetting only the upper soil layer may support the vegetation while the site is irrigated, but plants may not survive after irrigation ceases. Because roots grow preferentially into resource-rich sections of soil (Fitter and Hay 1987), surface watering will concentrate roots at the surface, reducing the deep root growth needed to sustain plants under drier conditions.

Although watering should soak the soil, excessive watering may cause erosion and/or nutrient leaching in areas with steep slopes or loose sediments. To promote soaking and reduce downslope runoff and erosion, it may be necessary to adjust watering to shorter but more frequent periods. Exact irrigation timing for a particular site will depend on fine-tuning a balance between maximum penetration of water and minimum erosion, as well as other factors discussed below.

Plants should be irrigated until they overcome their initial transplant shock. Irrigation should be maintained only until the vegetation has tapped into moist soil and additional watering is no longer required to maintain plant turgor; watering should not continue after the vegetation can survive on its own. Depending upon the season and conditions at planting, this period may last from one to a few months. Adequate root growth may take much longer if planting takes place during periods of natural dormancy or slow growth.

To wean plants from irrigation, the period of time between watering events should be increased gradually, while increasing the depth and duration of surface soil drying between soakings. This will encourage deeper root growth as the roots track the lowering soil moisture horizon (James and Zedler 2000). Reductions by 10 to 25% each week should allow well-established plants to acclimate to natural soil moisture and salinity. The length of time between watering events can be increased as long as the vegetation does not appear to be water stressed. Plant condition needs to be closely monitored while irrigation is being reduced, adjusting the duration and timing of soakings based on plant response.

Areas within a created wetland may have different irrigation requirements, and more than one watering scheme may be needed at a particular site. Sandier soils will dry more rapidly and deeply, especially when exposed to persistent winds. Higher elevations within the marsh that are well above the groundwater and tidal water inputs will initially require longer soaking periods. In addition, strategies for individual plant species differ, with some species being extremely tolerant of desiccation, while others rapidly wither and die without sufficient soil moisture. Species also differ in their seasonality, with most annual species germinating in winter and setting seed before the onset of desiccating summer

conditions, while perennials grow spring through summer to become dormant in late fall after producing seed.

Once irrigation has been terminated, the irrigation system should be left in place to be available for any plantings that need to be replaced. In addition, irrigation of the high-marsh may be necessary during especially dry periods in the early years of restoration. The need for irrigation may extend through three seasons or longer for areas that develop at a slow pace.

7.2.3 Exotic species invasions

Perhaps the greatest concern with prolonged irrigation, and the chief reason it should proceed no longer than absolutely necessary, is that freshwater irrigation can encourage the establishment of exotic, brackish, and freshwater species. Wetland-upland transitional habitats that are overwatered are readily dominated by exotic species that flourish during brief periods of low salinity and high moisture (Section 7.7.2; Noe 1999). The exotics can outcompete the slower growing native species that are adapted to drier and more saline conditions (Callaway and Zedler 1998). Once established, exotic species are often difficult or impossible to eradicate (Fellows 1999, Noe 1999). It is not obvious how to irrigate in order to promote native species establishment, while at the same time minimizing problems with exotic species. We recommend minimizing irrigation, with close monitoring of native species condition and exotic species establishment. Pulsing irrigation, with soaking periods alternating with surface drying, will favor natives that are adapted to spaced rainfall events. Intermittent drying between waterings should increase salt excretion by native halophytes, increasing the concentrations of salt on the leaves and within the plant; this may also have the beneficial effect of deterring some herbivores.

7.3 Replanting

Restoration projects often have patchy or low plant survivorship. In addition to natural, background plant mortality, mortality may be increased by poor planning, improper execution of plans, or stochastic events. Poor planning can result in low quality soils, ineffective hydrology, or insufficient spatial heterogeneity, all of which will reduce plant survivorship. In addition, each species persists best when planted in microhabitats for which it is adapted.

Plants are less likely to survive if they are too young, inadequately hardened to sun or salt, or arrive in poor condition due to an infertile growth medium, dehydration, or mechanical damage during handling. Some mortality will occur if planters break roots or stems, leave air pockets below roots, or compact soils. Little or no irrigation can also seriously reduce survivorship. Herbivores, human trampling, suffocating algal blooms, and invasive exotic species can all cause patches of high mortality. Assuming each species is planted in a suitable habitat with adequate care, stochastic events will affect plants once they are in the ground. Measures can be taken to eliminate or reduce many impacts (see below) but nothing will insure against all risks. Storms that cause extreme high tides or currents may erode planted areas or bury the restoration site in sediment. In southern California, hot dry winds off the desert occur unpredictably, and air humidity can plummet, stressing even vegetation that is deeply rooted. Such events are unpredictable and there is little that can be done to prevent their damages.

With so many potential causes of mortality, a budget should be set aside for replanting affected areas. Under the worst circumstances, corrective measures could require costly re-design and engineering.

7.3.1 Should you replant?

Some plant mortality is to be expected. However, with high levels of plant mortality (e.g., >50%), site development will be compromised, and the project may not achieve the restoration target on schedule. At this point, replanting may be desirable if the reasons for high mortality can be identified and overcome. However, if the site is simply not suitable for the species chosen, replanting will be a waste of resources, and the project will require re-design. High mortality over large portions of the site, in the absence of an extreme environmental event, indicates that the marsh has not been constructed or managed properly.

The decision to replant should be based on the likelihood that the cause of mortality will not recur. Catastrophic events can occur in successive years, but it is unlikely, so replanting after severe drought, erosion, or sedimentation is justifiable. In addition, replanting may be desirable and necessary when the original mortality was due to problems such as:

- transplanting species into unsuitable microhabitats, e.g., high-marsh species into lower marsh elevations;
- poor planting methods or conditions for establishment, e.g., weak plants, mechanical damage, or inadequate irrigation (as long as irrigation is not a substitute for providing the proper hydrology); and
- failure to protect sensitive transplants from predictable impacts that have been corrected, e.g., lack of fencing to keep out herbivores and people.

7.3.1.1 Timing

Reintroductions should be planned in advance and extra plants of all desired species kept on hand for replanting. The reintroduction of plants should be carefully timed to maximize growth potential under favorable conditions and to minimize damages before the plants have established (Section 4.3). This may mean waiting a full year to replant areas of heavy mortality. The Tidal Linkage marsh plain (Section 7.3.2) was replanted in early summer when extreme high tides were anticipated to inundate the marsh. Replanting before predictable harsh conditions is not advisable. Ideally, replanting should take place just prior to the time when plants break dormancy, as this will maximize the probability of survival (e.g., late winter in southern California). As always, we recommend that species diversity be maximized and reasonable care exercised in matching species with microhabitat, i.e., elevation and hydrology (Section 4.3.3).

7.3.1.2 Fertilizers

The use of fertilizers may be desirable to promote the growth of replacement transplants, especially if low nutrient soils contributed to the original transplant failure. However, even though fertilizers will promote greater initial productivity, nutrients may not be retained in sandy soils, and enhanced growth may not be maintained in subsequent years (Gibson et al. 1994, Boyer and Zedler 1999, Chapter 4). Incorporating fine, organic soil, similar to that in natural marshes and/or using organic amendments may improve soil quality. As a management tool, we recommend that fertilizers be used in addition to soil-conditioning amendments as a means of improving prospects for vigorous plant growth.

7.3.2 Case study: replanting at the Tidal Linkage

By replanting the seedlings that died at the Tidal Linkage (Box 1.10), we were able to maintain high overall survivorship and diversity in our experimental plots. Initial survivorship was

Table 7.1 Number of seedlings replaced between 15 May and 15 July 1997 at the Tidal Linkage experimental plots following mortality in spring and summer 1997. Plot totals represent the total number of plants for each species in the 87 2 × 2-m experimental plots. Seedlings were planted in a regular grid on 20-cm centers.

Species	Replaced	Plot Totals	% Survived	% Mortality
B. maritima	144	765	81.0	18.8
F. salina	5	780	99.4	0.6
J. carnosa	132	855	84.6	15.4
L. californicum	16	795	98.0	2.0
S. bigelovii	20	780	97.4	2.6
S. esteroa	18	810	97.8	2.2
S. virginica	6	885	99.3	0.7
T. concinna	133	810	83.6	16.4
Total	474	6480		
Mean			92.7	7.3

exceptionally high, with 93% of all plants surviving through the end of the first summer (Table 7.1). Approximately half of all the mortality was localized in one of five experimental areas, and most of this occurred in 3 of the 17 test plots in that area. Transplants were introduced on three separate occasions and were well established by the end of the summer. Based on visual inspection, survivorship was uniformly high in all test plots through the first fall (1997). However, a series of events affected all plants during the ENSO winter of 1997–98. Severe grazing by coots (Figure 7.2; Section 7.4.1), algal smothering (Figure 7.3;

Figure 7.2 Denuded marsh plain vegetation after coot herbivory at the Tidal Linkage restoration, February 1998. This level of damage was typical of all unfenced areas along channel banks (see Section 7.4.1).

Figure 7.3 Mats of algae (*Enteromorpha* spp.) overlying marsh vegetation in the lower marsh plain of the Tidal Linkage restoration, January 1998. Algal mats reduced light, retained water at low tide, and broke stems due to mechanical damage. Impacts were exacerbated when algal mats were later covered with sediment (see Figure 7.4).

Table 7.2 Number of seedlings replaced on 19 May and 16 June 1998 at the Tidal Linkage experimental plots. Mortality was associated with coot herbivory, algal smothering, and storm sedimentation during the winter of 1997–98. Plot totals represent the total number of plants for each species in the 87 experimental plots.

Species	Replaced	Plot Totals	% Survived	Mortality
B. maritima	90	765	88.2	11.8
F. salina	181	780	76.8	23.2
J. carnosa	66	855	92.3	7.7
L. californicum	200	795	74.8	25.2
S. bigelovii	56	780	92.8	7.2
S. esteroa	218	810	73.1	26.9
S. virginica	2	885	99.8	0.2
T. concinna	339	810	58.1	41.9
Total	1152	6480		
Mean			82.0	18.0

Section 7.5), and storm sedimentation collectively reduced cover and caused nearly 20% mortality among all plants (Table 7.2). Most of the mortality was concentrated in patches of intense impact at lower elevations (algal-smothered plants buried by up to 1.5 cm of sediment; see Figure 7.4.). Surviving plants increased in cover and canopy layering throughout 1998 (Keer 1999), and dead plants were replaced in summer 1998, as planned in the experimental design.

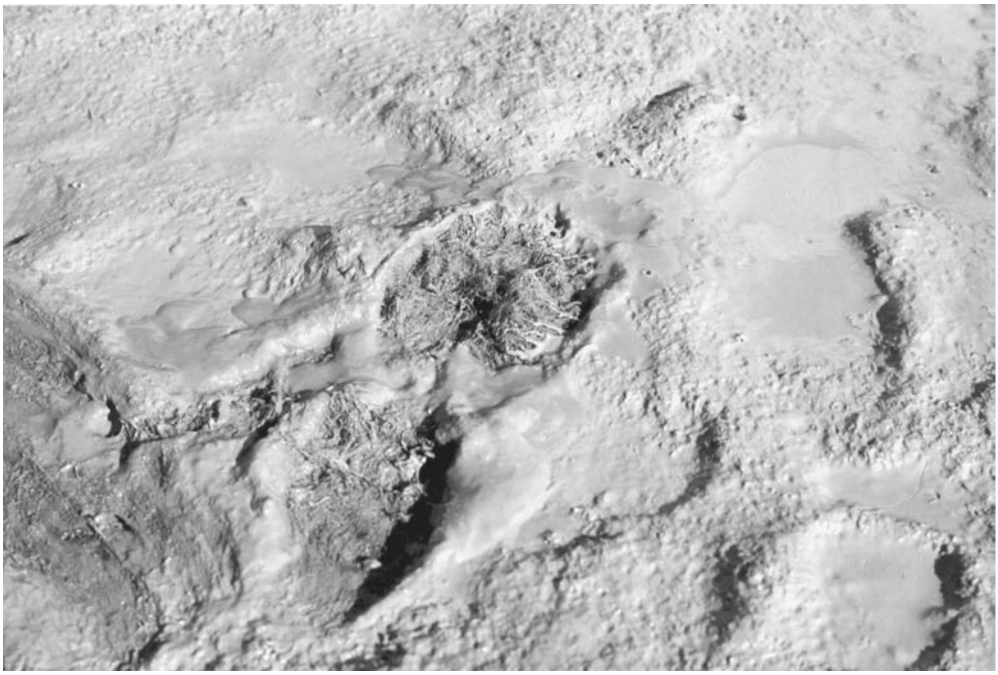

Figure 7.4 One to 1.5-cm of sediment from a single storm event blanketing a layer of *Enteromorpha* over marsh vegetation at the Tidal Linkage restoration site. Up to 10-cm of sediment was deposited on the vegetation near larger channels. Nearly all aboveground vegetation beneath the algae-sediment layer eventually died. There was little recovery of any species except *S. virginica*, which produced new shoots from rhizomes during the following summer.

Without replanting, most plots would still have achieved full cover through vegetative spread and recruitment, although with some loss in species diversity and some bare patches. *Limonium californicum* and *Frankenia salina* (two species most commonly found at higher elevations; Appendix 4) were damaged most at the low elevations and still did not perform well after replanting in these locations. *Triglochin concinna*, with the highest overall mortality (Table 7.2), grew poorly in monotype at all elevations, but established very well when planted in proximity to other species. The reasons behind this pattern of establishment were not clear, but some species do appear to establish best in mixed assemblages. Because patterns of mortality and subsequent recovery were related to species characteristics and the conditions at each plot, we underscore the importance of considering microhabitat preferences when introducing plants.

7.3.3 Case study: no replanting at Mission Bay (Crown Point)

An important benefit of replanting is the maintenance of plant species diversity. Where mortality is high and replanting is not pursued, vegetation may eventually be dominated by the most tolerant species. This is generally not the restoration objective.

At the Crown Point restoration project on Mission Bay (Box 1.5), we monitored survivorship among 9818 3-month old transplants. No measures were taken to promote establishment, and no replacements were made (see Section 4.4). Without irrigation, more than 33% of the transplants had died within 10 days, and more than 50% died in the higher elevation plots (Figure 4.9). Initial mortality appeared to be due to a combination of

dehydration and salinity stress. Overall survivorship leveled off at 23% among all plots after 10 months and was highest for *Salicornia virginica* at 36%. Ongoing mortality was species-specific and greatest in plots having the poorest drainage (maintaining standing water for extended periods at low tide; see Figure 4.2). After 18 months, diversity was severely reduced. Nine of the twelve plots were dominated by *S. virginica*, and eight of these were essentially monotypes of *S. virginica* (>99% relative cover). Despite close proximity to a natural source of propagules at the adjacent Kendall Frost Reserve (including the annual *Salicornia bigelovii*), there was little recruitment from any other species in plots dominated by *S. virginica*. Replanting can help counteract declining species richness and dominance by one or two tolerant species.

7.3.4 Recommendations

The history of restoration has shown that many projects will have problems with mortality associated with poor design, execution, or random events (Zedler 1988, Zedler 1994). Our experience suggests that replanting offers an opportunity to correct mistakes in the original planting design while enhancing development in the establishment phase. We recommend that replanting be considered part of an adaptive management strategy to promote the establishment of a diverse and functional wetland community. In developing a replanting strategy, managers should:

- first differentiate the causes of mortality, e.g., incorrect planting design, poor site conditions, poor planting methods, or catastrophic events;
- consider the patterns of growth and mortality of each species across the site before choosing when to replant;
- replant the impacted areas with species best suited to local conditions; and
- take corrective measures to reduce future impacts and enhance growth where appropriate.

7.4 Herbivory

Vertebrate and invertebrate consumer species are strongly attracted to a concentration of nutritious and palatable food (Mattson 1980, Nams et al. 1996). New vegetation at restoration sites often presents such a package of high-quality food, especially where restored sites are in close proximity to older habitat with a high concentration of animals. If food is scarce, as it is in late summer in our region, animals will likely be drawn to areas with newly establishing vegetation.

Transplants established with irrigation and/or at low soil salinity will have less salt on their leaf surfaces and lower salt concentrations in their tissues, potentially increasing palatability. Palatability may also be greater where new vegetation is significantly less lignified relative to plants found in older, perennial communities. In addition, the tissues of vegetation established with the aid of fertilizer, as is frequently the case at restoration sites, may have higher nutrient concentrations than that found in natural areas. Alternatively, where fertilizers are applied to infertile soils, such as commonly found at many restoration sites, plants may be released from nutritional stress, allowing them to allocate more resources to herbivore defense (Rhoades 1983, Boyer and Zedler 1996). Consequently, the use of irrigation and fertilizers to promote plant growth must be balanced against the potential loss to herbivores. This tradeoff provides a further rationale to limit irrigation and fertilization to the minimum levels needed to promote establishment and maintain growth (Sections 4.3 and 7.2).

Positive steps can be taken to reduce or eliminate the impact of herbivory so that growth far outweighs potential losses. In this section, we discuss how herbivory by birds, mammals, and insects has affected sites in San Diego County and elsewhere, and how these impacts may be limited. As with other management concerns, we recommend that steps be undertaken during the planning phase to minimize potential problems as the site develops.

7.4.1 Birds

Avian herbivores can inflict serious damage to new growth. Because of their ability to search from the air, birds can investigate a wide range of foraging areas, and the newly established vegetation at restoration sites is an easy target. Although there have been few studies examining avian impacts on restoration sites directly, a number of studies have shown that birds have significant impacts on salt marsh vegetation. Geese are frequently found grazing in wetlands where they dig for roots and rhizomes, killing plants and causing large soil disturbances in the process. Shifts in community composition from dicots to grasses to bare sediment may occur rapidly, with corresponding increases in salinity inhibiting recovery (Hik et al. 1992, Srivastava and Jeffries 1995, Miller et al. 1998). Ducks also forage on many aquatic and intertidal species, although their potential negative impact is less well documented.

Similarly, coots can be aggressive herbivores at both natural and restored sites. Coots are opportunistic generalists, naturally foraging throughout the coastal and inland wetlands of the western United States. They were responsible for extensive damage to transplants over the first winter at the Tidal Linkage restoration in San Diego, where they fed on the aboveground shoots of every available species except *Frankenia salina*, which was rejected for unknown reasons. Once discovered, steps were taken to keep coots out of the experimental plots, but they freely foraged over the rest of the site (see below).

Coots also had indirect effects through interactions with drift macroalgae. As a bloom of *Enteromorpha* blanketed much of the vegetation in the lower marsh elevations (Figure 7.3; Section 7.5), coots walked over the algal mat while foraging for protruding shoots. The weight of the coots on top of the saturated algal mat was enough to press all but the woodiest of plant parts into the soft surface sediments. Ongoing coot traffic drove shoots deeper into the sediment, which eventually became an anoxic soup of decomposing algae. Herbivory and anoxic sediment proved to be significant sources of mortality over the first winter. It is likely that nutrient deposition in the form of coot feces further served to fuel the *Enteromorpha* bloom and prolong its effects.

Because coots accessed the site by swimming to the channel edge and walking up to foraging areas, fencing was erected along the channel banks in the lower intertidal to keep them from the experimental plots (fencing had previously been erected across the high-marsh-upland transition zone; see below). Once at a site, the coots rarely flew and were only observed to take off and land on the water. The low fencing effectively kept the coots from gaining access to the experimental plots, even when these plots represented the only remaining vegetation at the restoration site. Although the coots could swim over the fence on the higher high tides, they did not stay within the exclosure as the tides receded. Where coots had unlimited site access, they stripped most plants of all non-lignified tissues in just a few weeks (Figure 7.2). The number of coots eventually decreased as more vegetation was lost, until the unprotected portion of the site was virtually bare by mid-March. Despite the intensity of this impact, the community slowly recovered the following summer from perennial rhizomes, but composition was markedly less diverse and the weediest species, *Salicornia virginica*, was dominant.

Fences were constructed of 60-cm tall "chicken wire" mesh cloth with 2.5-cm openings, a mesh that had little impact on hydrology, fish, or invertebrates. The fence was buried a few inches in the sediment to prevent access beneath it, and attached with nylon cable ties to 1-m tall wooden stakes spaced approximately 3 to 4 m apart. The fence maintained structural integrity for 2 years, after which the cable ties broke down. There was minimal accumulation of algae and other wrack material.

7.4.2 Mammals

Small mammals will likely present a significant herbivory problem at many restoration sites. In southern California, rabbits, ground squirrels, rats, and other terrestrial mammals move into the marsh during low tides and may feed extensively on marsh vegetation. These herbivores may also cause significant damage to plants being cultivated or stored temporarily onsite, or at nurseries prior to introduction. In our experience, they are selective foragers, preferentially choosing *Spartina foliosa, Triglochin concinna*, and *Suaeda esteroa*. At the Tidal Linkage, herbivory by rabbits caused significant damage to *S. foliosa* within weeks of its being planted, with some areas eaten to the ground. After the entire site was fenced, most of the remaining *S. foliosa* transplants recovered and did well. Despite its effectiveness against rabbits (the most common small mammal herbivore), fencing was less effective against other herbivores with the ability to climb, such as rats and squirrels.

In addition to terrestrial herbivores, aquatic species such as nutria or muskrats can cause extensive damage to wetland plants. Nutria especially have caused significant impacts in Gulf Coast wetlands, reducing vegetation up to 75% compared to fenced controls, with complete elimination of vegetation in some test plots (Taylor et al. 1997). Investigators found complex interactions taking place among herbivores, competitive effects, and abiotic stresses resulting in shifts in community composition, changes in dominant species, and decreased species diversity (Gough and Grace 1998*a,b*). Significantly, Taylor et al. (1997) suggest these herbivores tend to feed preferentially on more isolated and easily accessed clumps of vegetation, conditions commonly found in the early stages of a restoration project.

7.4.3 Insects

Insects may be the most difficult herbivores to control at restored wetlands, and little can be done to keep them away from host vegetation. Efforts that rely on pesticides to control phytophagous insects are generally unacceptable for most restoration projects as they kill target and valuable insects alike. Included among the beneficial insects are the predators needed to keep herbivores in check, as well as the salt marsh pollinators necessary for successful seed production. Conversely, phytophagous insect populations may be depressed in habitats supporting populations of their predators (Pfeiffer and Wiegert 1981). At the Connector Marsh in San Diego, Boyer and Zedler (1996) found that fertilization enhanced cordgrass growth and reduced the scale insect infestation, possibly by attracting greater numbers of predatory beetles to the taller fertilized plants.

Because of the detrimental effects of pesticide use and the beneficial effects of creating an intact and functional community, we recommend an integrated pest management approach similar to that used in agricultural systems. This means recognizing that phytophagous insects are a natural part of the marsh ecosystem, and that management efforts be aimed at reducing their impact rather than eliminating their presence. Primarily, this means developing a plant community that is able to withstand natural levels of herbivory. As suggested throughout this volume, the site should be designed to promote the production

of healthy plants within a heterogeneous landscape with appropriate hydrology and adequate soil quality. Maximizing plant diversity on a local scale will reduce the chance for entire areas to be affected by specialist herbivores, while maintaining marsh functioning through the compensatory growth of less palatable species. Additionally, target plants will be less apparent when growing within a matrix of nontarget species. Habitats with just one native species (i.e., the lowest marsh elevations with pure *S. foliosa*) are an exception to the "maximize diversity" rule.

7.4.4 Recommendations

A number of options can be pursued to decrease the probability of serious vegetation losses due to herbivory. Depending upon the size and scope of the project, fencing represents an effective deterrent to vertebrate herbivores, as well other nuisance species, such as dogs and cats. However, despite its effectiveness against coots and rabbits, fencing cannot exclude all herbivores. Ducks, squirrels, rats, and larger digging or chewing herbivores, such as nutria, will not be foiled by short fences.

At larger sites and in places with complex topography, fencing should be integrated with other approaches. Efforts to control herbivory by geese have often relied upon scaring the birds away, using cannons or other loud devices to keep the birds from landing and grazing. A program of live trapping may also be effective against long-term resident herbivores such as coots and squirrels. More lethal solutions, such as snap traps or poison baits, are less desirable because they kill unintended targets; hence, they are generally not permitted.

We recommend that managers:

- develop a response plan as part of the overall adaptive management strategy,
- monitor the site so that the plan can be implemented before serious damage occurs,
- plant a locally diverse mix of species at the smallest scale, avoiding dense monocultures whenever possible (Chapter 4),
- erect fencing to keep herbivores out until the site has developed sufficiently to withstand impacts (recommend a minimum of 2 years),
- prepare deterrents to foraging vertebrates, such as warding noises or raptor decoys,
- trap nuisance herbivores that do not respond to deterrents,
- avoid pesticides, except as a last resort to avoid total failure over a large area (if permitted), and
- establish healthy vigorous plants that can withstand impacts, while providing suitable habitat for an entire invertebrate community, including predators.

7.5 Macroalgal blooms

Macroalgae occur naturally in coastal and estuarine ecosystems throughout the world. Green macroalgae of the genera *Ulva, Enteromorpha, Chaetomorpha*, and *Cladophora* are key components of salt marsh creeks and mudflats, often driving primary production, light relationships, and nutrient dynamics (Zedler 1980, 1982; Owens and Stewart 1982; Fong and Zedler 1993; Fong et al. 1993*b*, 1996, 1997; Kwak and Zedler 1997; Hauxwell et al. 1998). Blooms can cause significant and devastating impacts to mudflats dominated by seagrasses or invertebrates by changing the physical and chemical environment, such as light penetration and dissolved oxygen concentrations (e.g., Rosenberg et al. 1990, den Hartog 1994). Macroalgal blooms also may affect structure and cover of the vascular plant canopy, alter competitive relationships among seedlings, and limit plant dispersal or survivorship (Soulsby et al. 1982, Everett 1994, Norkko and Bonsdorf 1996, Raffaelli et al. 1998).

Severe macroalgal blooms are commonly driven by eutrophication associated with coastal development (Fong et al. 1993*a*, Bombelli and Lenzi 1996, Short and Burdick 1996, Short and Wyllie-Echeverria 1996). Based on observations in southern California and northwestern Mexico, such blooms play an important role in salt marsh dynamics. When blooms occur in lagoon or mudflat habitats, the tides move large mats of drifting macroalgae onto the marsh vegetation, where it is entrained as the tides recede. Extensive accumulations may crush and smother the underlying vegetation, creating opportunities for seedling recruitment in otherwise closed communities dominated by long-lived perennials. In addition, some algal species grow at lower elevations in the salt marsh, attached to vegetation or directly on the sediment under relatively open canopies. As in mudflat habitats, the decomposition of imported or locally grown algae represents a potentially important source of nutrients to the salt marsh system (Fong et al. 1993*c*, 1994).

7.5.1 Potential impacts at restoration sites

Severe algal blooms can affect restoration efforts in several ways. During periods of productive growth, drifting macroalgae may settle on and entangle salt marsh plants. In addition, macroalgae within or under the canopy may accumulate enough biomass to blanket vascular plants, strongly reducing the light environment and inducing anoxic conditions where overlying mats press vegetation to the ground (Figure 7.3). Such algal mats have been responsible for extensive seedling mortality among desirable species naturally establishing at restoration sites (see Figure 7.5). Dense mats of algae also retain water at low tides, so that relatively level areas drain slowly or not at all. Sediment carried on tides may accumulate on macroalgal mats, and the subsequent burial of the organic material (Figure 7.4) can quickly promote hypoxia and the potential buildup of toxic

Figure 7.5 Macroalgal impacts to seedling recruitment at the Tidal Linkage, March 1998. The low density of *salicornia bigelovii* seedlings in the lower part of the photo occurred where an algal mat buried emergent seedlings during the winter of 1997–98. High seedling densities in upper portion of the photo occurred just outside the mat.

sulfide levels. Nutrients released from reduced sediments and decomposing algae or plant material may then promote more macroalgal growth, further reducing light and oxygen levels beneath the overlying mat (Raffaelli et al. 1998). Sensitive transplants can die even with relatively short exposure to reduced light, anoxia, and chronic inundation. All of these impacts were noted at the Tidal Linkage, especially after coots grazed on the algae and trampled plants into the sediments (Section 7.4.1). If macroalgae accumulate over well-established plants, recovery is more likely (Figure 7.5).

Algal blooms are a natural part of the salt marsh environment, and little can be done to reduce their impact. Once blooms occur, removing volumes of saturated algae is generally not possible and would be extremely damaging to vegetation where it is entangled and growing on plants. The preferred management strategy should be to control nutrient inputs, especially from sewage spills and fertilizer run-off. Areas that recover poorly might need to be replanted (Section 7.3).

7.6 Sedimentation issues

Sediment input is a key factor affecting the elevation of the marsh. As sediment accumulates, elevation increases and fewer tides reach the new marsh surface. Because of these processes, the marsh plain elevation tends to stabilize around MHW (Section 3.2.2). Because relative elevation is a primary determinant of vegetation composition, changes in sediment dynamics directly affect the development of restored salt marshes.

For large areas of the U.S. coast, the primary concern related to elevation and sediment dynamics has been submergence (DeLaune et al. 1983, Baumann et al. 1984, Stevenson et al. 1985, Kearney and Stevenson 1991). The nation's largest areas of wetland loss occur in Louisiana, where accretion cannot keep pace with subsidence and rising sea level (Boesch et al. 1994). More subtle imbalances in sedimentation, subsidence, and rising sea level will lead to shifts in salt marsh communities (Warren and Niering 1993). However, where local sediment supplies have increased due to disturbances within coastal watersheds and where subsidence is not a problem, the concern is just the opposite: excessive sedimentation converts one desired habitat to another type. This is the case in southern California, where some mudflats are shifting to cordgrass marsh and marsh-plain areas are building to high-marsh habitats or even beyond the tidal range. Although inputs of tidal sediment decrease as elevation increases over the marsh surface, inputs of storm sediments are not limited to areas influenced by the tides. In areas where local watersheds drain directly into wetlands, storms deliver even greater volumes of sediment. Storm-related sediments significantly alter elevation and vegetation within wetlands (Section 7.6.3), and large areas of former channels and marsh can be shifted to upland elevations (Zedler et al. 1992).

7.6.1 Target elevations: the strategy of overexcavation and dredge spoil consolidation

Early restoration sites were designed to have a static elevation, and the wetland was excavated to the target elevation, as determined from a natural, reference wetland. Over the last decade, it has become clear that a more dynamic design is appropriate, incorporating plans for the evolution of the geomorphology via overexcavation and the accumulation of sediments (Williams *in press*). However, it remains uncertain how much overexcavation and evolution is appropriate for a particular location or habitat. Two questions need further refinement for designing future projects: (1) How quickly will the site accumulate sediments? (2) How quickly will dredge spoil material consolidate, i.e., how much will it become compacted?

To decide how much overexcavation is appropriate for a particular site, background data are needed for:

- natural rates of sedimentation in nearby reference marshes,
- concentrations of suspended sediment in the water column,
- the time period desired for plant establishment, and
- the stability of unconsolidated sediments.

Overexcavation is most appropriate in areas with high suspended sediment loads (Section 3.2.6). Where rates of sedimentation and suspended sediment loads are low, it may take a long time for the excavated elevations to build up. Further research is needed to fine-tune these methods for particular locations and to identify more clearly the expected time frame for site development.

Overexcavation is common for establishing creeks within a restored marsh, and we recommend this practice. Excavation to depths approximately 10% below the target allows creeks to evolve and accumulate sediment of the proper texture for benthic invertebrates. Because of their frequent tidal inundation, creek bottoms should evolve relatively rapidly. It is much safer to err on the side of overexcavating creeks than underexcavating, because deposition will occur more rapidly than erosion (Williams 1986, Coats et al. 1995). If creeks are too shallow, tidal flushing of the site will be restricted, slowing the progress of restoration.

For habitats constructed with a slurry of dredge-spoil material, the design elevation may be well above the target elevation, anticipating the consolidation of spoils pumped onto a site (Section 3.2.6.1). Where newly deposited dredge spoils will be deep, it can be difficult to estimate the amount of consolidation that will take place. Because vegetation is sensitive to differences in elevation of a few centimeters above or below the target, it is essential to achieve precise elevations. Furthermore, dredged material is very difficult to modify once in place because tractors are easily mired in soft material. Given these problems, there is a real need for more controlled studies of consolidation of dredged material. Estimates of consolidation rates are currently being developed at restoration sites in the Mississippi River Delta Plain (Andy Nyman, *personal communication*).

7.6.2 Sediment dynamics in the early phases of restoration

Sediments are likely to be dynamic immediately after a restored system is opened to tidal action. Tidal hydrology drives sediment dynamics, eroding some areas and depositing sediment in others. Initial changes can be very large if there is a poor match between the morphology of the site and the tidal channels that are excavated. Even after the site has reached a relatively stable topography, small-scale evolution of tidal creeks and the marsh plain will continue. During the initial evolution of the site, the following should be tracked:

- erosion and slumping of channel banks in areas with sparse vegetation,
- changes along slope banks at the edge of the marsh-upland interface, and
- excessive sediment accumulation and burial of new plants.

Spartina spp. and other plants that grow at the lowest elevations in the marsh may be able to stabilize sediments along creek banks and mudflats. In the absence of vegetation, sediments are much more likely to be resuspended and slopes to erode. Denser planting along creek banks should help to stabilize soils in these erosion-prone areas. Similarly, denser plantings may be needed on the slopes of other transitional areas (e.g., from high-marsh to upland habitats) to prevent slumping or other shifts in morphology.

7.6.3 Storm sediment impacts

Storm events generate large-scale movements of sediment, and extensive deposits can bury vegetation and alter elevations by 1 to 10 cm, enough to shift plant species composition. Even greater inputs occur under the most extreme conditions (Section 3.2.5.1). Watersheds with steep topography, erodible soils, and highly variable rainfall are particularly susceptible to storm-pulsed sedimentation, as are small watersheds that drain immediately and directly into coastal wetlands. A single pulse of sediment can shift marsh elevation out of the tidal range.

In addition to raising elevation, pulsed inputs of sediment can smother vegetation. In the south arm of Tijuana Estuary, the 1995 flood deposited 10 to 30 cm of sandy sediment along a small tidal channel. *Salicornia virginica* tolerated burial by up to 20 cm of sediment by growing through the new deposit (Figure 3.6; Callaway and Zedler in preparation). Other species' tolerances have not been observed, but wetland-upland transition areas with elevated high marshes (e.g., Sweetwater Marsh, San Dieguito Lagoon) have few perennial species. *Salicornia virginica* and *S. subterminalis* predominate where storm sediment inputs are apparent (areas with increased elevations adjacent to steep watersheds). Their persistence, in place of upland species invasions, may be due to the pumping of salts from buried marsh soils to the surface.

Storm-related impacts are unavoidable in both natural and restored wetlands. With this in mind, restoration sites should be designed for resiliency, but expectations should not exceed those for natural systems. That is, we should expect impacts while striving to create sites that are as resilient to catastrophic events as are natural wetlands (Box 2.4). This requires that we position the wetland where it will be protected from fluvial inputs and dune washover. A landscape-scale planning approach is thus called for.

7.6.4 Management implications

We have emphasized the need to locate the restoration site appropriately within the watershed and estuary and to achieve the proper hydrology. After vegetation is established, further grading or excavation would damage both plants and animals. Therefore, the planning and implementation phases are critical.

Preliminary analyses or modeling of hydrology and sediment dynamics should help compare alternative sites for the restoration project. Issues to consider include the range of sedimentation rates in nearby natural reference sites, the suspended sediment load in adjacent tidal channels, and the degree of channel stability in reference sites. Furthermore, there should be space at the restoration site for the hydrology of the system to develop. Channels should have room for migration, and buffers around the site should be large enough to allow natural shifts in habitats over time. Current project designs for wetland restoration frequently lack broad buffers between wetland and upland areas because upland buffer acreages do not earn mitigation credit, i.e., they are not considered to compensate for loss of other wetland habitat. The lack of buffer habitats limits the flexibility of restoration sites to develop, especially when the development of the wetland is not easily predicted.

The watershed and landscape contexts for the restored site should indicate the best location for the restoration site, namely, an area that is not likely to receive direct inputs of sediment during storm events. The condition of the watershed immediately upstream of the restoration site should be assessed, and additional hydrologic and sedimentation modifications planned where slopes are highly disturbed. The position of the restoration site relative to marine sediment sources (e.g., proximity to unstable dunes) should also be considered. In cases where a restoration project cannot be moved out of a high-risk

location, berms surrounding the site or sediment basins upstream may help protect the site. Vegetated berms are likely to be more stable and also provide some habitat values. However, the use of sediment basins and extensive berms will require ongoing maintenance and should be avoided where possible.

Finally, all restored wetlands should be monitored for sediment accumulation. Even if sediment dynamics cannot be altered, the findings will contribute to a better understanding of sedimentation processes, which will in turn generate better designs of future projects.

7.7 Exotic plant invasions

Exotic plants are a common problem at many restoration sites where soils are disturbed and plant cover is initially low. If native plants are not quickly established, exotics can gain a foothold and outcompete the target native species. Given the high likelihood that exotic plants will invade at least the higher elevations of the restoration site, steps should be taken to minimize their establishment and spread.

Control of exotics requires an understanding of their life history. Successful exotic species tend to have relatively high production of propagules, broad dispersal, high rates of establishment and rapid growth, and many spread by vegetative methods (Baker 1965, Bazzaz 1986, Rejmanek and Richardson 1996). The most noxious exotic species have many of these strategies. Most species are limited either by dispersal or require disturbed substrates for establishment; dispersal and disturbance limitations are two ends of a spectrum, with many species in between. In addition, each successful invasion is the result of an interaction between the invading species, the native species present, the habitat, and the environmental conditions at a site (Crawley 1987), so all of these factors must be considered in controlling the exotic species. Perrins et al. (1992) note that management practices at a site should also be considered in analyzing the speed of exotics.

7.7.1 Dispersal-limited species

Dispersal-limited species have propagules that are unlikely to travel long distances. These species typically become established after intentional plantings (*Spartina* spp.) or by escaping from horticultural stock (*Limonium* spp.). There are few dispersal-limited species in our coastal wetlands; however, they can be the most problematic. Once they appear nearby, they are available to invade undisturbed natural habitats or restoration sites.

The dispersal-limited species that are most problematic for coastal wetlands are species of *Spartina* (Figure 7.6). Within this genus, numerous species have infested wetlands around the world: *Spartina alterniflora* and *Spartina anglica* colonize bare mudflats, and *Spartina densiflora* and *Spartina patens* invade the marsh plain. *Spartina alterniflora* has been introduced to the Pacific Northwest (Frenkel 1987), San Francisco Bay (Callaway and Josselyn 1992), New Zealand, (Partridge 1987), and other locations. After introduction to Europe, it hybridized with the native *Spartina maritima*, eventually resulting in the polyploid hybrid, *S. anglica*, which is now the most common low-marsh species in Europe (Thompson 1991). *Spartina anglica* has been intentionally planted for erosion control and land reclamation worldwide, with extensive efforts in China and elsewhere (Chung 1983, 1993). In some areas, there are concerns that the colonization of mudflats by *Spartina* spp. restricts shorebird feeding in these areas (Goss-Custard and Moser 1988). In the Pacific Northwest, there is concern that oyster aquaculture areas and local fisheries will be damaged as *S. alterniflora* expands in Willapa Bay and *Spartina anglica* in Puget Sound (Daehler and Strong 1996). Chemical and mechanical efforts to control these *Spartina* species have shown only limited success, while biological control has shown some promise

Figure 7.6 Native *Spartina foliosa* (right) and the exotic *Spartina alterniflora* (left) growing together in south San Francisco Bay. The introduced species is almost twice as tall as the native species. Since being intentionally introduced, *S. alternilfora* has spread rapidly at both natural and restored sites in this bay. It is an example of a dispersal-limited species (Section 7.7.1).

to control invasive cordgrass in areas without native *Spartina* species (Wu et al. 1999). In San Francisco Bay, *S. alterniflora* has hybridized with the native species, *Spartina foliosa*. The hybrid produces prolific seed (Ayres et al. 1999), and newly restored areas in south San Francisco Bay are much more likely to be colonized by the exotic or hybrid species than the native. Regional goals for wetlands in San Francisco Bay identify this as a significant problem for restoration efforts in south San Francisco Bay (Goals Project 1999).

Once established, dispersal-limited species can be extremely difficult to eradicate because, by definition, the habitat is suitable for them. Hence, the best control measure is to prevent them from establishing or to remove them shortly after they invade the restoration site. Removal may require pulling, digging, smothering, or applying herbicides, all of which have significant negative impacts on other species in the wetland. Regular monitoring and rapid eradication of new patches of exotics are needed. Without frequent monitoring, aggressive species can rapidly dominate. Once an unwanted species covers a large area, its removal may be impossible or so damaging that this offsets the benefits of removal.

7.7.2 Disturbance-limited species

The grading or excavation of restoration sites creates ideal conditions for disturbance-limited species to establish. Such species thrive where there is little vegetation cover or where the soil surface is disrupted. In some cases, the disturbance that facilitates invasion is more subtle, e.g., increased flood frequency, nutrient loading, or altered soil salinity. Eradication efforts thus need to address the environmental alterations responsible for invasion, in addition to their removal after invasion.

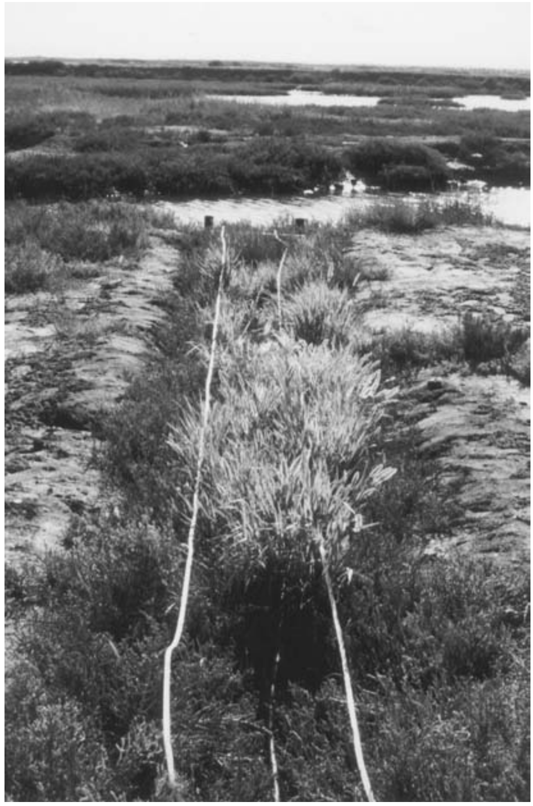

Figure 7.7 An experimental mesocosm at Tijuana Estuary became dominated by an exotic grass, *Polypogon monspeliensis*, a disturbance-limited species (Section 7.7.2).

In southern California coastal wetlands, the increase in year-round freshwater inputs, which reduces the salinity of both water and soil, is a major cause of exotic plant invasions. Examples include *Parapholis incurva*, *Polypogon monspeliensis* (Figure 7.7), and *Mesembryanthemum crystallinum*. The hydrologic changes and the exotic invaders are most noticeable in transitional or marginal wetland areas, i.e., places with less common tidal inundation than the marsh plain. Marsh areas that are regularly flooded by the tides have few disturbance-limited exotic species because frequent tidal inundation sustains high salinity. However, if freshwater inflows are prolonged, brackish and freshwater marsh species (e.g., *Scirpus* spp. and *Typha* spp.) can establish in salt marshes (Beare and Zedler 1987). *Rumex crispus* and *Scirpus robustus* are good indicators of excess freshwater influence.

Although limited in their establishment ability, disturbance-limited species tolerate a wide range of environmental conditions and are likely to persist well after the evidence of disturbance disappears. Prevention and early control are thus advised.

7.7.3 Potential impacts of exotic plants

Exotic plants are of concern at restored wetlands (and natural areas) because they alter ecosystem functioning. Potential impacts are:

- competition with native species and loss of species diversity,
- hybridization with native species,
- shifts in habitat structure and distribution,

- changes in primary productivity and food web dynamics, and
- changes in hydrology and sediment dynamics.

Stuart Findlay (*personal communication*) is exploring changes in nutrient cycling and food quality where purple loosestrife (*Lythrum salicaria*) and *Phragmites australis* have invaded tidal freshwater wetlands of the Hudson River. Although the impacts of exotic plants in coastal wetlands are not thoroughly studied, changes in nitrogen dynamics (Vitousek and Walker 1989), habitat use (Trammel and Butler 1995), and other impacts (D'Antonio and Vitousek 1992) have been documented in other habitats.

7.7.4 Management implications

For dispersal-limited species, such as *Spartina* spp., monitoring and prevention are the best strategies. If these species cannot be prevented from establishing, they will be difficult to remove from restoration sites and natural wetlands. Introduced *Spartina* species can alter habitats and could cause substantial problems for some restoration projects. For disturbance-limited species, we need to address the disturbance that has allowed them to proliferate. We need to establish environmental conditions at the site that favor growth and spread of native species over exotics. Irrigation should be minimized, as excess freshwater can favor disturbance-limited exotics (Section 7.2). Fertilization of native vegetation should be carefully considered as well because disturbance-limited exotics may be favored. For both disturbance-limited and dispersal-limited exotic species, removal of the problem species early after invasion is a high priority.

7.8 Exotic animal species

As with exotic plants, many exotic animals thrive under the disturbed conditions that are found at restoration sites. These species also are extremely difficult to control or eradicate once they are well established. In the last decade, extensive research has evaluated the impacts and spread of estuarine invertebrates (Ruiz et al. 1997, Carlton and Geller 1993). Although most of this work has focused on benthic invertebrates in deep water habitats, rather than wetland species, many of the findings and recommendations apply to wetlands in general and restoration projects in particular. Many species are dispersal-limited and have been established with the discharge of ballast water from ships (Ruiz et al. 1997). Wetlands near shipping ports are thus particularly vulnerable.

7.8.1 Problem species of invertebrates and fish and their impacts

The primary impact of exotic animals is their direct competitive effect on native species. In addition, some species cause ecosystem-level changes to natural habitats. The burrowing isopod, *Sphaeroma quoyanum*, has been found in San Francisco Bay wetlands since the 1800s, and it is now abundant throughout the estuary (Carlton 1979, Josselyn 1983). It is also common in southern California wetlands (Levin et al. 1999). *Sphaeroma quoyanum* burrows into channel banks, and as burrow densities increase, channel banks are weakened, causing slumping. This is of special concern in restored wetlands where vegetation is sparse and channel banks are particularly susceptible to slumping. Levin et al. (1999) found that erosion rates along creek banks with *S. quoyanum* burrows were highly variable but averaged 8 cm in San Diego Bay and 18 cm in San Francisco Bay over a 6-month period.

Musculista senhousia also has significant impacts on ecosystems where it becomes abundant. This small mussel is abundant in both Mission Bay and San Diego Bay, and it is found in small numbers in Tijuana Estuary. It commonly occurs within beds of *Zostera*

marina. This mussel impacts abundances of native benthic invertebrates by building dense mats that increase the physical complexity of the substrate. As a result, many species increase in abundance; however, planktonic developers can be reduced (Crooks 1998, Crooks and Khim 1999). *Musculista senhousia* affects beds of *Z. marina* by reducing rhizome elongation rates, especially where *Z. marina* is patchy and sparse, indicating that this species may have significant impacts in newly restored eelgrass beds (Reusch and Williams 1998).

In addition to invertebrates, many introduced fish species have become established in coastal wetlands and are of concern for restoration projects. Unlike invertebrates, exotic fish do not usually modify habitats where they are introduced; however, they can affect native fish assemblages through competition with native species and shifts in food web dynamics. The yellowfin goby (*Acanthogobius flavimanus*) is one of the most abundant introduced species in California wetlands. This predator was first found in San Francisco Bay in the 1960s and reached San Diego County in the 1980s. Populations in San Diego Bay wetlands are dominated by young-of-the-year, indicating that wetland habitats serve as nursery areas for these introduced species (Williams et al. 1998). Williams et al. (1998) also documented the occurrence of an uncommon exotic species, the sailfin molly, in San Diego Bay wetlands. Meng et al. (1994) found a decline in native fish species abundances in Suisun Marsh over a 14-year period that was associated with changes in freshwater inflow, salinity, and the abundance of exotic fish species, including the yellowfin goby and the chameleon goby.

The best method for controlling exotic animals is to prevent their establishment, primarily through more stringent control of ballast water dumping. Once exotic animals are widespread at a site, they are extremely difficult to remove because of their high reproduction rates, mobility, and ease of reestablishment. In addition, more disturbed habitats are more likely susceptible to invasion. In freshwater streams, Baltz and Moyle (1993) found that undisturbed streams with abundant native fish populations are able to resist invasions by introduced fish species.

7.9 Human activities

7.9.1 Access issues

There are both potential benefits and impacts of facilitating human access to a restoration site. Human appreciation, involvement, and understanding of restoration projects is highly desirable; at the same time visitors can have negative impacts on plants and animals. Depending on the overall goals of a particular restoration site, an appropriate balance between visitor access and protection of resources should be established. Restoration sites offer outstanding educational opportunities, and human involvement in the restoration process is a valuable way to reestablish human connections with natural systems (Cowell 1993). Educational opportunities should receive high priority for access because the public's increased understanding of habitat values will benefit future restoration efforts.

Public access and high visibility opens restoration sites to vandalism of monitoring equipment, signs, or habitats themselves. Care should be taken to hide, secure, and disguise monitoring equipment so that it is not easily visible, and does not appear to be of high value. We recommend locked containers. In addition to vandalism, excess noise, movement, and lights along trails will affect wildlife use of the habitat. Josselyn et al. (1989) found lower bird use in areas with high human use, although some birds acclimate to human disturbances. Impacts of public access deserve further study, as there is a great need to determine how buffers can be designed to reduce the impacts of visitors on wildlife.

Along with human access, cats and dogs can have significant impacts on natural and restored wetlands. Predation by cats (both domesticated and feral) can affect resident bird populations and other wildlife, although it is not clear how significant this impact may be (Patronek 1998). Efforts should be made to reduce access of cats to restoration sites in order to reduce negative impacts to wildlife populations. In addition, dogs can trample vegetation, disturb birds, and significantly alter wildlife use if they are allowed easy access to restored areas. Proper design of the site to minimize impacts from cats and dogs and education of the public concerning these impacts are necessary preventatives.

7.9.2 Pollutants: nutrients, toxic materials, and trash

Because of their location at the base of watersheds, coastal wetlands are frequently impacted by stormwater runoff and wastewater inputs (treated or untreated). Increased nutrient inputs are likely to cause algal blooms or shifts in the relative abundance of primary producers (Section 7.5). Toxic materials and biohazards can have impacts on restored sites, especially when they are located in urbanized areas. Accumulations of metals and organic pollutants in estuarine and wetland sediments have been evaluated in a variety of different areas (Valette-Silver 1993), but impacts of these pollutants on vegetation are not obvious (Williams et al. 1994). Bioaccumulation of pollutants in the food chain is a concern; however, there has been little research to evaluate potential impacts of these pollutants at restored wetlands. The best remedy for these problems is pollution prevention and control.

In addition to nutrient and pollutant impacts in urban areas, debris can float into coastal marshes via stormwater and tidal inundation. In restored wetlands in San Diego County, we have encountered a wide range of trash, including telephone poles, garden trimmings, tires, and parts of broken boats and docks. The volume is often large enough to bury vegetation. In some cases, it may be necessary to install a trash boom to prevent debris from accumulating in the wetland. Prevention of the accumulation of pollutants is the best alternative, as cleanup efforts will require additional access and trampling of sites.

7.10 Summary

The development of restoration sites is difficult to predict, and unexpected events can occur as each site progresses. Given the uncertainty, it is advantageous to prepare for and minimize potential problems during the planning and design phase. The ideal for restoration projects is to become self-sustainable ecosystems, that is, sites that need minimal management. In the planning phase, special consideration should be given to:

- locating the restored or created wetland so that it best fits the landscape,
- scheduling irrigation so it will enhance transplant survival but minimize exotic plant invasions,
- designing plantings so that individual species are matched to their preferred microsites,
- preparing a response strategy in advance for unpredictable impacts, such as herbivory, algal blooms, exotic invasions, or sedimentation events,
- monitoring development so that an adaptive response can be implemented in time to limit negative impacts,
- planning ahead in case replanting is needed, and
- planning access so that humans can appreciate the project but not have negative impacts.

Maintenance problems can be minimized by careful monitoring of restoration sites as they develop. If problems are identified when remedies are still possible, modifications can be minimal. Monitoring should include evaluations of algal blooms, herbivory, sediment dynamics, and exotic species.

Finally, maintenance problems, mid-course corrections, and the outcomes of various corrective measures should be recorded and reported in the literature, so that others can learn from the experience. Where possible and practical, experimental approaches should be used in trying mid-course corrections, so that the best methods for future restoration projects will be developed quickly.

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chapter eight

Conclusions and future directions

Joy B. Zedler

8.1 Overview

The overriding challenge for coastal wetland restoration is to conserve and sustain biodiversity and ecosystem functions. As of December 1999, we cannot guarantee that our efforts to manage coastal wetlands will have this desired effect. The status of the art and science of restoration is rudimentary, relative to the diversity and complexity of wetland ecosystems. At the same time, there is enormous promise that research and adaptive restoration approaches will rise to meet the challenge.

To date, restoration efforts have demonstrated that:

- It is possible to expand the area of coastal wetland through a number of procedures, such as removing fill, breaching dikes, halting drainage, and renewing connections with flowing water.
- Renewing the wetness of coastal habitat has the potential to conduct some ecosystem services and attract some native species.
- Not all species or ecosystem functions are easily restored.
- There is a high degree of individuality among the species' responses to restoration efforts; the same is likely true about ecosystem processes, although these are less well documented.
- Attempts to restore species or functions often have surprise outcomes.
- Every project can offer new information and new understanding about how to restore coastal wetlands — if the site is properly monitored, the data interpreted, and findings disseminated.
- Restoration outcomes depend on at least two major variables: the degree of degradation of the site and the type of actions taken to restore it (Chapter 1).
- We are far from being able to predict the outcome of most restoration actions that might be taken at any of a range of different restoration settings. We cannot yet guarantee that, if I take "action x" at "site y," the outcome will be "result z."
- By considering where in the "restoration spectrum" (Chapter 1) each restoration effort fits, and by carefully documenting how the site progresses over time, we could gain that predictability ("action x, done at site y, will produce result z").

In agriculture, range management, landscape architecture, and horticulture, there are long traditions of working out ways to achieve specific results, often with just a few

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species, but in a moderate range of environmental settings. The number of scientists and practitioners involved in bringing these respective fields to their present-day maturity is very large. In contrast, it is expecting a great deal of the newer field of restoration ecology to be able to produce whole ecosystems of many types in a very broad range of environmental settings, some of which are totally new to the landscape (e.g., cat-clays that develop when salt marsh soils are exposed to the air). Restoration has not been practiced by many people or for very long periods of time. That restoration ecology — the science — is immature is obvious in the simplicity of the models available for predicting the pathways and outcomes of the practice. We have a long way to go, but we have come a long way, too.

Aldo Leopold was one of the first professional restorationists in the US. In 1934, he dedicated the University of Wisconsin-Madison Arboretum to the task of reconstructing examples of native Wisconsin communities on a 485-ha farm that was purchased by the university to become a research and educational facility. He and other biologists, including John Curtis, Norman Fassett, and Henry Greene, then set about replanting a horse pasture to native grassland. They knew little of how to accomplish this feat, but they knew a lot about which species were available for reintroduction. They collected seeds and transplants from remnant prairies and began placing them in the pasture. Fire experiments by Curtis and Partch (1948) led to controlled burning regimes designed to favor native species over exotic pasture grasses and weeds. About 170 native plant species now occupy the site.

In 65 years, a great deal has been learned about how to restore prairies, and the information has been transferred to and tailored for hundreds of other sites within the region. Courses in restoration ecology have appeared on the curricula of universities; graduate students have begun to conduct their thesis research on restoration sites; annual conferences have been scheduled to disseminate findings; prairie nurseries have come into being; consulting firms have arisen to meet demands for implementing prairie restorations; citizen groups have developed to help landowners restore prairies; programs for creating schoolyard prairies have "propagated" across the state of Wisconsin; and home gardeners have adopted native prairie species as either horticultural plantings or alternatives to bluegrass (*Poa pratensis*). The Arboretum's Curtis Prairie serves as a regional (and perhaps national) model or demonstration site; it shows that restoration efforts can prevail, that native species can be reintroduced, and that the experience of visiting a native prairie can be mimicked. Over the past 65 years, prairie restoration has evolved to the state where one can obtain recommendations for what to do ("action x") in different places ("site y") in order to expect at least the dominant grasses and many forbs to establish ("result z;" see Packard and Mutel 1997). Still, prairie restorationists cannot guarantee the desired outcomes for all restoration settings, and few restored prairies match the diversity of prairie remnants. Common complaints are that forbs are outcompeted by aggressive native grasses, insect diversity is low (mostly generalists), native mammals are missing (specialists), and grassland birds fail to use restoration sites. There is still much to be learned.

Coastal wetland restoration lags well behind prairie restoration efforts. Plans to restore tidal wetlands are being developed and implemented throughout the country, from the northeast south to Florida and around the Gulf of Mexico, and from the Pacific Northwest to southern California. But each region needs its own demonstration sites (Zedler and Weller 1989) and its own handbook for restoration. In this book, we have described several examples of attempts to achieve specific outcomes. In no case can we promise certain results. But in the process of trying to restore species and habitats and processes, we learn more about how to improve our efforts. It is on this optimistic note that we wish to leave our readers.

8.2 Conclusions

From Chapter 1, we conclude that restoration theory has shortcomings and is ripe for improvement. Simple "one size fits all" models are unlikely to withstand field testing; it is more reasonable to expect similarity in responses — and predictability — within cells of a restoration spectrum, where the axes relate to the type of degradation and the actions taken. There is much to be done in testing for generalities and, if they develop, in filling such a spectrum. An adaptive restoration approach, incorporating large experiments into restoration sites — or designing the restoration site as an experiment — is warranted.

From Chapter 2, we conclude that goal setting is a critical phase that is best when tailored to the site, using historical records and contemporary information from reference sites. It is equally important to consider the views of people in the planning process, so that restoration efforts will have local support. The final restoration model should incorporate spatial heterogeneity, in order to support biodiversity, recognize temporal variability, and be realistic.

From Chapter 3, we conclude that hydrology is the forcing function of wetlands, and restoring the complex structure and functioning of coastal wetlands depends on restoring hydrological processes. The substrate from which wetland soil is expected to develop must be of the appropriate texture to function in nutrient supply and water retention or drainage. Once vegetation is in place, these features cannot be easily altered, so initial plans should rely on scientific information.

From Chapter 4, we conclude that native plant species decline when tidal flushing is impaired, and that reintroducing lost species can be difficult, with many constraints and surprises. Experimental plantings are very helpful in deciding what species to plant and where, how plants and soils with seed banks can be salvaged, what types of propagules work best, how plants should be hardened before planting, when plants should be introduced to the site, when irrigation and fertilizers are useful, and how to ensure that planted sites have adequate genetic diversity. Unlike agriculture and horticulture, the desire is to maximize diversity and build in the potential for change over time. Vegetation is restorable where the hydrology and soils are adequate.

From Chapter 5, we conclude that fishes and invertebrates are best restored by mimicking the topography and hydrology of tidal creek networks. Because these components of the wetland are spatially and temporally variable, fishes and invertebrates are associated with a variety of habitats within tidal wetlands, including subtidal and intertidal areas. Habitat heterogeneity and connectivity are critical for the functioning of fish and invertebrate assemblages. Creating a square subtidal basin with no marsh connection will not maximize species diversity, food support, or sustainability.

From Chapter 6, we conclude that *adequate* assessment requires not only a systematic and long-term approach to monitoring, it also requires a coordinated research program to explain the patterns that emerge from the data. Recognizing that our experience in conducting simultaneous monitoring and research programs is unusual, we provide a guide to methods for sampling physical and biological features of wetlands and suggest what efforts might constitute a minimal monitoring program.

From Chapter 7, we conclude that tidal wetland restoration sites will need maintenance, and adequate budgets should be provided for short- and long-term actions, such as irrigation, replanting, fertilizing, and discouraging herbivores and other pests. To be able to deal with algal blooms, sedimentation events, and exotic plant invasions requires continual observation and an ability to respond to unexpected events.

Overall, it is wiser and more efficient to protect wetlands from degradation, rather than to damage them and try to compensate for the losses in structure and function. Another southern California example demonstrates this principle: in planning to transport treated wastewater from the City of Tijuana to a new ocean outfall, the Environmental Protection Agency and other agencies decided to tunnel underground, so that the 3.6-m diameter pipeline could pass 45 m beneath Tijuana Estuary, rather than through the salt marsh (surface excavation). This critical — and appropriate — decision kept a 60-m wide swath of Tijuana Estuary from being damaged. Consistent with federal mitigation policy that damage to wetlands be avoided wherever possible, this bold project is a model for the future. We look forward to the day when avoidance is always the choice and when restoration can focus on repairing historical degradation. And we look even further to the future when restoration has been completed, all wetlands have been restored and are self-sustaining, and handbooks such as this one have no further utility.

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Appendix 1

Native and nonnative salt marsh plant species of San Diego County

Gary Sullivan

Native Species

1.	*Amblyopappus pusillus* Hook. and Arn.	A	Pineapple weed
2.	*Atriplex californica* Moquin.	P	California saltbush
3.	*Atriplex triangularis* Willd.	A	Spear-leaved saltbush
	A. patula L. ssp. *hastata* Hall and Clements misapplied		
4.	*Atriplex watsonii* Nelson	P	Matscale
5.	*Batis maritima* L.	P	Saltwort
6.	*Cordylanthus maritimus* ssp. *maritimus* Benth.	A	Salt marsh bird's-beak
7.	*Cressa truxillensis* Kunth	P	Alkali weed
8.	*Cuscuta salina* Engelm.	P	Dodder
9.	*Distichlis spicata* (L.) Greene	P	Salt grass
10.	*Eriogonum fasciculatum* Torrey and Gray	P	California buckwheat
11.	*Frankenia palmerii* Watson	P	Palmer's frankenia
12.	*Frankenia salina* (Molina) Johnston	P	Alkali heath
	formerly F. grandifolia *Cham. and Schldl.*		
13.	*Heliotropium curassavicum* L.	P	Seaside heliotrope
14.	*Hutchinsia procumbens* (L.) Desv.	A	Hutchinsia
15.	*Isocoma menziesii* var. *vernonioides* (Nutt.) Nesom	P	Coast goldenbush
	formerly *Haplopappus venetus* ssp. *vernonioides* (Nutt.) Hall		
16.	*Jaumea carnosa* (Less.) Gray	P	Jaumea; Seaside daisy
17.	*Juncus acutus* L.	P	Spiny rush
18.	*Juncus bufonius* L.	A	Common toad rush
19.	*Lasthenia glabrata* L. ssp. *coulteri* Gray	A	Coulter's goldfields
20.	*Limonium californicum* (Boise) Heller	P	Sea lavender
21.	*Lycium californicum* Nutt.	P	Box thorn
22.	*Monanthochloe littoralis* Engelm.	P	Shoregrass
23.	*Ruppia maritima* L.	P	Ditch-grass; Widgeon grass
24.	*Salicornia bigelovii* Torrey	A	Annual pickleweed
25.	*Salicornia europaea* L.	A	
26.	*Salicornia subterminalis* Parish	P	Glasswort
	also *Arthrocnemum subterminale* Ferren		
27.	*Salicornia virginica* L.	P	Pickleweed
	also *Sarcocornia v.* Ferren; formerly *S. pacifica*		
28.	*Spartina foliosa* Trin.	P	California cordgrass

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29.	*Spergularia macrotheca* (Hornem.) Heynh.	P	Large flowered sand-spurry
30.	*Spergularia marina* (L.) Griseb.	A	Sand-spurry
31.	*Suaeda calceoliformis* (Hook.) Moquin.	A	Horned sea-blite
32.	*Suaeda esteroa* Ferren and Whitmore	P	Sea-blite
33.	*Suaeda moquinii* (Torrey) Greene	P	Bush seepweed
34.	*Suaeda taxifolia* Standley	P	Wooly sea-blite
35.	*Triglochin concinna* Burtt Davy	P	Arrow-grass
36.	*Zostera marina* L.	P	Eelgrass

Nonnative species

1.	*Atriplex semibaccata* R.Br.	P	Australian saltbush
2.	*Bassia hyssopifolia* (Pallas) Kuntze	A	Hyssop-leaved bassia
3.	*Cotula coronopifolia* L.	A	Brass buttons
4.	*Limonium ramosissimum* ssp. *provinciale* Pignatti	P	Provincial sea lavender
5.	*Lolium multiflorum* Lam.	A	Italian rye grass
6.	*Lythrum hyssopifolium* L.	A	Hyssop's loosestrife
7.	*Mesembryanthemum crystallinum* L.	A	Crystalline iceplant
8.	*Mesembryanthemum nodiflorum* L.	A	Slender-leaved iceplant
9.	*Parapholis incurva* Hubb.	A	Sickle grass
10.	*Polypogon monspeliensis* L.	A	Rabbit's-foot grass
11.	*Rumex crispus* L.	P	Curlydock
12.	*Sonchus asper* L.	A	Prickly sow thistle
13.	*Sonchus oleraceus* L.	A	Common sow thistle

A = annual; P = perennial.

Appendix 2

Coastal wetland plant species of southern California

Gary Sullivan and Gregory B. Noe

Introduction

This appendix will assist in the identification, collection, and propagation of coastal wetland plants of southern California for use in restoration. It is not a key. We include descriptions of 36 native species found in the salt marshes and salt marsh transitional habitats of San Diego County. We also include descriptions of the 13 most commonly found nonnative species to encourage their identification and management. Because the transition from salt marsh to upland or freshwater wetland forms a zone of overlapping species ranges rather than a distinct border, we include the transitional species consistently associated with the native halophytes. Although some of these have broad upland distributions, they are important to the functioning of salt marsh ecosystems. Provided in this appendix are:

- species identity and classification, following Hickman (1993; *The Jepson Manual*), listing recent taxonomic changes, conventions used by others, family, names in common usage, and the U.S. Fish and Wildlife Service wetland rating;
- categorization as native or nonnative, annual or perennial, with a short description of growth form and scale to accompany species drawings;
- type and color of flowers or inflorescences, with pollinators, if known;
- primary mode of reproduction (not listed for annuals);
- range and median elevations in feet relative to mean low low water (MLLW), and meters relative to 0.0 m NGVD 29 (see Appendix 4 for the references or sources of elevation ranges);
- species phenology, including timing of germination, growth, flowering and fruiting, and the window of opportunity for seeds to be collected; and
- restoration notes, including how seeds may be collected and stored, how plants may be propagated, how they establish naturally, and other relevant considerations.

For nonnative species, we have omitted information on how to facilitate cultivation and substituted information that may facilitate management efforts.

Descriptions of growth form, flowering, and mode of reproduction are based on personal observations and may differ slightly from published keys. Our observations

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reflect the flora of San Diego County unless otherwise noted. For example, *Spergularia marina* is listed as an annual, but it persists where soil moisture is available throughout the summer and fall.

Elevation range and median are provided as a guide in placing species in appropriate habitats. However, elevation is a simple surrogate for the complex interactions between hydrology, soil, and other environmental characteristics that determine plant survival, growth, and reproductive capability (Zedler et al. 1999). Plants respond to the effect that flooding and drainage patterns have on salinity, oxygen, water, and nutrient availability, and indirectly to the effects these factors have on the species with which they interact. Species distributions result from these complex interactions, so that the elevation profile of each species may differ slightly among different microhabitats and entire salt marsh systems. For example, since creek banks drain relatively fast, they support many of the species adapted to the less saturated soils found at higher elevations, in addition to those found on the adjacent flats. Undercut creek banks slump over in time, lowering the elevation cross section of surviving species, which may be facilitated by ongoing clonal integration with ramets found in adjacent, higher areas. Consequently, it is best to:

- measure species' actual elevation distributions where restorations take place in proximity to an existing marsh;
- plant species most densely around median elevations, and less densely around the upper and lower range limits; and
- consider microhabitat characteristics in addition to elevation when devising a planting scheme.

The elevation data presented here and in Appendix 4 were taken from global position system or autolevel measurements relativized to known benchmarks by Sullivan or Noe (*unpublished data*). Benchmark elevations were taken or verified with a global positioning system courtesy of either Bruce Nyden or Michelle Cordrey from PERL. Corroborating elevations were measured by Zedler (1977), Brewster (1996), Bradshaw (1997), Trnka (1998), Fellows (1999), James (1999), Zedler et al. (1999), and Ward (2000). Some species elevations were not relativized to a benchmark but were estimated from their position relative to other species (see Appendix 4).

Descriptive and phenological observations were taken by Sullivan and Noe, with additional input from Meghan Fellows (*Cordylanthus maritimus* ssp. *maritimus, Distichlis spicata, Monanthochloe littoralis*), Matt James (*Lycium californicum*), and Sally Trnka (*Spartina foliosa*).

San Diego County salt marshes are designated by the following abbreviations:

TJE (Tijuana Estuary National Estuarine Research Reserve)
ORE (Otay River Estuary)
SBA (South Bay Biological Study Area)
SWM (Sweetwater Marsh National Wildlife Refuge)
SDB (San Diego Bay)
FS (Famosa Slough)
SDR (San Diego River)
KFR (Kendall-Frost Reserve)
LPL (Los Peñasquitos Lagoon)
SDL (San Dieguito Lagoon)
SEL (San Elijo Lagoon)
BL (Batiquitos Lagoon)
AHL (Agua Hedionda Lagoon)

BVL (Buena Vista Lagoon)
SLR (San Luis Rey River)
SMR (Santa Margarita River)

Native species

Amblyopappus pusillus Hook. and Arn. (Asteraceae). **Pineapple weed.** FACW-[1]

Native annual forb; erect stem; 3 to 20 cm tall.
Flowers: yellow composites, 3-mm diameter; insect pollinated.
Elevation range: 6.8 to 8.1 feet (median 7.5) MLLW; 1.3 to 1.7 m NGVD.

Phenology

Germination: Nov–Mar, dependent upon seasonal rainfall.
Shoot production: germination — May.
Flowering: Mar–Jun.
Seed collection: Apr–Jul.

Restoration notes

Seed collection, processing, and storage: seeds are sticky and should be stored separate from other species.
Propagation techniques: an annual species, seeds germinate in moist, low salinity soil.
Field establishment: readily germinates and establishes in response to the increased soil moisture and decreased salinity following significant rainfall.
Special considerations: a relatively drought-tolerant species found in seasonally dry transitional habitats.

Found in San Diego County at TJE, ORE, SBA, SWM, LPL, AHL, BVL, SMR

Atriplex californica Moquin. (Chenopodiaceae). **California saltbush.** FAC

Native perennial forb, subshrub; decumbent-erect stems, to 25 cm, spreading to >50 cm wide; lanceolate/elliptic leaves, 5 to 24 mm.
Flowers: monoecious; female flowers in axils of leaves, male inflorescence terminal.
Reproduction: sexual
Elevation range: 5.8+ feet MLLW; 1.0+ m NGVD.

Phenology has not been studied here.

Restoration notes

Special considerations: an inconspicuous species, perennating from underground rhizomes. Found in both upland marginal habitat and in high marsh with *M. littoralis, S. subterminalis, C. truxillensis,* and *P. incurva.*

Found in San Diego County at TJE, SWM, LPL

Atriplex triangularis Willd.; *A. patula* L. ssp. *hastata* (L.) Hall and Clements has been misapplied (Chenopodiaceae). **Spearscale.** FACW

Native annual forb; ascending/erect shoots 10 to 50 cm; triangular-hastate leaves to 70 mm.
Flowers: monoecious; spike staminate (terminal) and pistillate inflorescences; thought to be wind pollinated.
Elevation range: 5.5+ feet MLLW; 0.9+ m NGVD.

Phenology

Germination: Mar–May.
Shoot/leaf production: germination–Aug.
Flowering: Jul–Sep.
Seed collection: Sep–Oct.

Restoration notes

Seed collection, processing, and storage: collect seed as flowers mature, store at cool temperatures.

Propagation techniques: readily propagated from seed.

Field establishment: germinates in late spring after other high salt marsh annual species.

Special considerations: found at variable elevations where there is freshwater influence.

Found in San Diego County at TJE, ORE, SWM, FS, SDR, KFR, LPL, SDL, BL, AHL, SLR, SMR

Atriplex watsonii Nelson (Chenopodiaceae). **Matscale.** FACW+

Native short-lived perennial; prostrate stems forming dense, white-scaly mats, 5 to 20 cm high, <2 m dia.; elliptic/ovate leaves to 25 mm.

Flowers: spiked staminate and pistillate inflorescences, wind pollinated.

Reproduction: sexual; dioecious.

Elevational range: 6.2 to 9.5 feet (median 7.2) MLLW; 1.1 to 2.1 m NGVD.

Phenology

Germination: Dec–Feb, depending upon rainfall.

Shoot/leaf production: Jan–Dec.

Flowering: Mar–Jul.

Seed collection: Jun–Sep.

Restoration notes

Seed collection, processing, and storage: timing of seed production needs to be determined.

Propagation techniques: readily propagated from seed. Cuttings root well.

Field establishment: readily germinates and establishes in field.

Special considerations: a short-lived perennial species that seems to prefer relatively open canopy.

Found in San Diego County at TJE, SBA, SWM, SDR, KFR, LPL

Batis maritima L. (Bataceae). **Saltwort.** OBL

Native perennial forb; decumbent or ascending stems rooting at nodes, with upright flowering stems 15 to 30 cm tall; cylindrical succulent leaves, 1.5 to 3 cm long.

Flowers: yellow-green inflorescences to 1 cm, in the axils of leaves; fruits to 3 cm; thought to be wind-pollinated.

Reproduction: primarily asexual from new shoots produced at the nodes of rhizomes; dioecious.

Elevation range: 4.2 to 7.8 feet (median 5.4) MLLW; 0.5 to 1.6 m NGVD.

Phenology

Germination: Feb–Apr.

Shoot/leaf production: Feb–Sep.

Flowering: Jul–Oct.

Seed collection: Sep–Nov (best Oct–early Nov).

Restoration notes

Seed collection, processing, and storage: seed may be collected as the fruits mature, turning from green to white with the look and texture of popcorn. Dried fruits should fragment easily. Refrigerated seed has remained viable at PERL for 2+ years.

Propagation techniques: seed germination is low, but after germination seedlings grow well. Cuttings establish very well, rooting in water or soil after 3 to 4 weeks.

Field establishment: natural germination is generally poor and varies among sites. In restoration sites, seedlings establish well, with survivorship enhanced by early irrigation. Cuttings may also be planted directly into the marsh at elevations without prolonged periods between flooding, using young herbaceous stems with approximately 5 cm exposed above ground.

Special considerations: seed viability differs greatly among sites; germination seems to have been enhanced by dormancy or cold storage longer than 2 to 3 months.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR

Cordylanthus maritimus ssp. *maritimus* Benth. (Scrophulariaceae). **Salt marsh bird's-beak.** OBL

Native annual hemi-parasite; an erect forb developing dense ascending flowering branches; 5 to 15 cm tall, to 40 cm wide.

Flowers: white to purple flowers, 3 to 5 mm diameter, 1 to 2 cm long; bee pollinated.

Elevational range: 5.0 to 9.7 feet (median 6.8) MLLW; 0.76 to 2.2 m NGVD.

Phenology

Germination: Nov–Apr, depending upon rainfall.

Shoot production: germination–Aug.

Flowering: May–Aug.

Seed collection: Jul–Sep.

Restoration notes

Seed collection, processing, and storage: a rare and endangered species, seeds may not be collected without permits.

Propagation techniques: soaking seeds in water 24 to 72 hours prior to planting may improve germinability.

Field establishment: readily germinates and establishes in canopy gaps with low salinity.

Special considerations: this is a state and federally-listed endangered species. As a hemiparasite, this species will not establish well without appropriate host species (primarily *M. littoralis*; Fellows 1999).

Found in San Diego County at TJE, SWM

Cressa truxillensis Kunth (Convolvulaceae). **Alkali weed.** FACW

Native perennial forb; erect stems from rhizomes; 5 to 20 cm tall.

Flowers: white, five sepal flowers, 3 to 5 mm dia.

Reproduction: primarily asexual from rhizome (seedlings not observed, and may not be readily distinguishable from other shoots).

Elevational range: 5.8 to 9.9 feet (median 7.7) MLLW; 0.99 to 2.24 m NGVD.

Phenology

Germination: has not been studied here.

Shoot/leaf production: new shoots produced from Dec–May.

Flowering: May–Aug.

Seed collection: May–Aug.

Restoration notes

Seed collection, processing, and storage: collect seed from mature flowers anytime.

Propagation techniques: establishes readily from plugs with rhizomes.

Field establishment: grows readily from seed in disturbed areas.

Special considerations: may be relatively dense in open areas, or inconspicuously found among dense clonal perennials, especially *M. littoralis*.

Found in San Diego County at TJE, ORE, SBA, SWM, LPL, SDL, SEL, BL, AHL, SMR

Cuscuta salina Engelm. (Cuscutaceae). **Dodder**. NLR

Native annual or facultative perennial, an obligate parasitic forb; liana growing on other plants, stems may form a dense twining anastomosis; <1 mm diameter.

Flowers: white, 2 to 5 mm diameter, 5 lobed; insect pollinated.

Reproduction: sexual.

Elevational range: 4.5 to 7.5 feet (median 5.9) MLLW; 0.6 to 1.5 m NGVD.

Phenology

Germination: Feb–Mar (may be longer, observations are few).

Shoot/leaf production: nearly year round, varying among patches within sites.

Flowering: Mar–Jan, varying among patches within sites.

Seed collection: as patches mature seed, highly variable.

Restoration notes

Seed collection, processing, and storage: collect seeds from mature flowers, up to 4 seeds per flower.

Propagation techniques: establishes well from seed in the greenhouse when germinating with hosts.

Field establishment: plants establish from seed. The stem immediately seeks out a host plant. Soon after haustoria formation, stem connection with the soil is abandoned.

Special considerations: non-photosynthetic (orange) obligate parasite in marsh and upland habitats. It may cause host mortality locally. Found most commonly in the salt marsh on *F. salina* and *S. virginica*, and less so on other species, such as *J. carnosa* or *T. concinna.* It appears to germinate both in the soil and the canopy. Grows and flowers nearly year-round in areas that remain moist.

Found in San Diego County at TJE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL

Distichlis spicata (L.) Greene (Poaceae). **Saltgrass.** FACW

Native perennial grass; decumbent to erect culms, 10 to 50 cm tall, branching from stolons; new shoots arise at the nodes of rhizomes.

Flowers: inflorescence spikes from green to purple, 2 to 8 cm; wind pollinated.

Reproduction: primarily asexual; dioecious.

Elevational range: 4.9+ feet MLLW; 0.7+ m NGVD.

Phenology

Germination: seedlings observed in April, no data on germination.

Shoot/leaf production: Apr–Nov.

Flowering: Apr–Oct.

Seed collection: Jun–Oct.

Restoration notes

Seed collection, processing, and storage: seeds collected in late summer, dried and stored in a cool dry place.

Propagation techniques: establishes readily from rhizome cuttings including a node.

Field establishment: new shoots arise primarily from rhizomes. Establishes very well in restoration sites from plugs, rooted cuttings, bare root shoots, rhizome fragments with nodes, or nursery plants.

Special considerations: plants lie dormant between Dec and Feb. An aggressive species in the high marsh-transitional areas that may locally occur well onto the mid-marsh plain. Also provides important habitat for the rare salt marsh wandering skipper (*Panaquino panoquinoides*). Spreading quickly in year 2, it provides

good ground cover — especially on slopes. A superior competitor in sandier high marsh and upland transitional habitats.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Eriogonum fasciculatum (Benth.) Torrey and Gray (Polygonaceae). **California buckwheat.** NLR

Native perennial shrub; decumbent to ascending shoots; 60 to 120 cm tall, 50 to 100 cm wide.

Flowers: white-pinkish clusters, 3-mm diameter, in cyme-like inflorescence; pollination probably by bees.

Reproduction: sexual.

Elevation range: 7.8+ feet MLLW; 1.6+ m NGVD.

Phenology

Germination: has not been observed.

Shoot/leaf production: year round, but primarily Dec–Apr.

Flowering: Mar–Jun.

Seed collection: May–Aug (best Jun–Jul).

Restoration notes

Seed collection, processing, and storage: collect inflorescences as they begin to turn brown, seeds may be separated or left in the flowers; store flowers or seeds in cool dry place.

Propagation techniques: seeds germinate well in flats. We have not used cuttings here, although they may establish well and could be tried.

Field establishment: naturally establishes from seed, it can be introduced to restoration sites as seedlings or possibly cuttings.

Special considerations: found at lower elevations than most other species growing in the high marsh — upland transition zone; to increase the likelihood that salt tolerant genotypes will be used in restorations, use seed or plant material collected from the margins of nearby coastal wetlands.

Found in San Diego County at TJE, SWM, FS, KFR[2,] LPL, SDL, SEL, BL, AHL

Frankenia palmerii Watson (Frankeniaceae). **Palmer's frankenia.** OBL

Native perennial shrub; woody convoluted stems forming a dense dome, 20 to 50 cm high; leaves <10 mm, curling as the soil dries.

Flowers: white 5 petaled flowers, 5 to 10 mm dia. in the axils of leaves at terminal branch ends; insect-pollinated.

Reproduction: sexual, we have not observed evidence of asexual reproduction.

Elevation range: 7.0+ feet MLLW; 1.35+ m NGVD.

Phenology

Germination: following rainfall, Dec–Mar.

Shoot/leaf production: Feb–Aug.

Flowering: May–Jul.

Seed collection: Jul–Sep.

Restoration notes

Seed collection, processing, and storage: seeds remain in the flower until the shoots or flowers senesce. Seeds can be harvested by collecting mature flowers, each with a variable number of seeds. Flowers or seeds should be air dried and stored in cool temperatures.

Propagation techniques: plants grow well from seedlings or rooted cuttings. Cuttings root in moist sandy soil.

Field establishment: natural establishment may be rare in this long-lived and slow growing species. Seedling establishment is probably rare under dense canopy cover. In restoration sites, it has established well from rooted cuttings.

Special considerations: a very rare species, the only natural population in the U.S. is adjacent to the Chula Vista Nature Interpretive Center at Sweetwater Marsh (plus introduced from cuttings to two other locations). It is widely distributed around the salt marshes of Baja California.

Found in San Diego County at TJE, SWM, KFR

Frankenia salina (Molina) Johnston; formerly *F. grandifolia* Cham. and Schldl. (Frankeniaceae). **Alkali heath.** FACW+

Native perennial subshrub; ascending/upright stems from rhizomes; forms dense cover, 10 to 40 cm high.

Flowers: small 5 petal flowers, 5 to 12 mm dia, white, pink, or bluish purple; in the axils of leaves at terminal branch ends; insect-pollinated.

Reproduction: primarily asexual with shoots developing from perennating rhizomes, with new populations establishing from seed.

Elevational range: 5.2 to 7.5 feet (median 6.3) MLLW; 0.8 to 1.5 m NGVD.

Phenology

Germination: Jan–May.

Shoot/leaf production: Feb–Sep.

Flowering: Apr–Nov.

Seed collection: Sep–Dec (best Oct–Nov).

Restoration notes

Seed collection, processing, and storage: seeds remain in the flower until the shoots or flowers senesce. The seeds can be harvested by collecting mature flowers, each with a variable number of viable seeds depending upon pollination success. Flowers or seeds should be dried and stored in cool temperatures. Seeds last up to 2+ years.

Propagation techniques: plants grow well from seedlings or rooted cuttings. Cuttings root quickly in water or moist sandy soil. Seeds germinate quickly with freshwater, often with multiple plants developing from each flower head if seeds are not separated out.

Field establishment: plants establish naturally with new shoots arising at the nodes of rhizomes; seedling establishment is probably rare in mature marsh. In restoration sites, may establish very well from seedlings, plugs, rooted cuttings, or nursery plants.

Special considerations: germination promoted by lowered salinity and higher temperatures in spring. Longevity of stems responds to water availability, with higher elevation plants senescing earlier as soils dry out.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SMR

Heliotropium curassavicum L. (Boraginaceae). **Seaside heliotrope**. Native perennial forb; OBL

Growth form: prostrate — ascending stems, 3 to 12 cm tall, spreading to <80 cm.

Elevational range: 6.8+ feet MLLW; 1.3+ m NGVD.

Reproduction: sexual.

Flowers: 2 to 4 terminal inflorescence spikes, ends coiled while in flower; flowers 3 to 5 mm with white corolla.

Phenology has not been studied here.
Found in San Diego County at TJE, SWM, SDR, KFR, LPL, SEL, BL, SLR, SMR

Hutchinsia procumbens (L.) Desv. (Brassicaceae). **Hutchinsia**. Native annual forb; NLR
Growth form: erect forb; 1 to 10 cm tall.
Elevation range: 7.2 to 8.1 feet (median 7.4) MLLW; 1.4 to 1.7 m NGVD.
Flowers: small white flowers, 1-mm diameter.
Phenology
Germination: Nov–Mar, depending upon rainfall.
Shoot production: germination–Apr.
Flowering: Jan–Apr.
Seed collection: Jan–Apr.
Restoration notes
Seed collection, processing, and storage: seed capsules quickly dehisce and seeds should be collected soon after plant death.
Propagation techniques: annual species, field establishment suggested.
Field establishment: readily germinates and establishes in areas with low salinity.
Special considerations: an extremely ephemeral and often overlooked species.
Found in San Diego County at SWM, SDL

Isocoma menziesii (Hook and Arn.) Nesom var. *vernonioides* (Nutt.) Nesom; formerly *Haplopappus venetus* ssp. *vernonioides* (Nutt.) Hall (Asteraceae). **Coast goldenbush.** NLR
Native perennial subshrub; decumbent to erect shoots; 50 to 120 cm tall, 20 to 40 cm wide; leaves with toothed margins.
Flowers: yellow heads in terminal cymes; insect pollinated.
Reproduction: sexual.
Elevation range: 8.0+ feet MLLW; 1.7+ m NGVD.
Phenology
Germination: not observed.
Shoot/leaf production: Apr–Dec.
Flowering: summer–fall.
Seed collection: Sep–Nov (best Oct–Nov).
Restoration notes
Seed collection, processing, and storage: collect seeds while akenes are golden in color, seeds are usually eaten by the time akenes turn brown; store in cool dry place.
Propagation techniques: seeds germinate well in flats; cuttings root well.
Field establishment: naturally establishes from seed, it can be introduced to restoration sites as seedlings or cuttings.
Special considerations: *Isocoma* grows lower in elevation than most other upland species. For coastal wetland restoration, collect seeds from plants growing around wetland margins. Shoots are green in summer when most other upland species are dormant. Found in coastal sage scrub surrounding most coastal wetlands.
Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SMR

Jaumea carnosa (Less.) Gray (Asteraceae). **Jaumea; Saltmarsh daisy.** OBL
Native perennial forb; prostrate, ranging from sparse understory stems to dense monotypic mats, 10 to 20 cm high.
Flowers: yellow composites, 2-cm diameter; insect-pollinated.

Reproduction: primarily asexual from shoots rooting at the nodes of stolons, although extensive reproduction from seed may occur in response to freshwater flooding and lowered salinity.

Elevation range: 4.5 to 7.0 feet (median 5.5) MLLW; 0.6 to 1.35 m NGVD.

Phenology

Germination: Feb–Apr.

Shoot/leaf production: Feb–Oct.

Flowering: May–Sep.

Seed collection: Jun–Sep (best Aug; varies locally).

Restoration notes

Seed collection, processing, and storage: seed ripens precociously, may be collected while ripe swollen fruits are still green, or shortly after ripening from the dried fruit — after which seed is shed rapidly. Dried seed should be stored in cool temperatures, best if used within one year.

Propagation techniques: seeds germinate readily in moist soil. Cuttings root very quickly and easily in water or soil.

Field establishment: plants establish naturally from new shoots rooting at the nodes of stolons, with seedling establishment probably rare in mature marsh. In restoration sites, establishes very well from seedlings, plugs, rooted cuttings, or nursery plants. Cuttings may also be planted directly into the marsh at elevations without prolonged periods between flooding, using young herbaceous stems with approximately 5 cm exposed above ground.

Special considerations: an important matrix species at mid elevations, it does extremely well in areas with freshwater influence.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Juncus acutus L. (Juncaceae). **Spiny rush.** FACW

Native perennial subshrub/shrub; cespitose acicular leaves, sharp stiff spines to 1.5 m.

Flowers: 2 to 4 mm flowers, petals rounded, 6 anthers in lateral inflorescences.

Reproduction: sexual, and asexually from nodes on the multi-branched rhizomes.

Elevation range: 7.5+ feet MLLW; 1.5+ m NGVD.

Phenology has not been studied here.

Restoration notes

Propagation: grows readily from seed, germinating well under moderate salinities (up to 10 ppt; Zedler and Beare 1986). Clones can be dug entire and transplanted.

Field establishment: readily establishes from seed after flooding events.

Special considerations: found in and around salt marsh margins, lower in elevation in areas receiving freshwater influence. May be an ideal species to form natural barriers around the margins of restoration projects to keep out humans and nuisance species.

Found in San Diego County at TJE, SWM, FS, SDR, LPL, SDL, SEL, BL, BVL, SLR, SMR

Juncus bufonius L. (Juncaceae). **Toad rush.** FACW+

Native annual graminoid; erect culms; 1 to 7 cm tall.

Flowers: small green to white flowers, 1 to 2 mm diameter.

Elevation range: 7.2+ ft MLLW; 1.4+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.

Shoot production: germination–Apr.

Flowering: Feb–Apr.
Seed collection: Mar–May.
Restoration notes
Seed collection, processing, and storage: seed capsules quickly dehisce and seeds should be collected soon after plant death. Tiny seeds can be collected by shaking the mature flowers.
Propagation techniques: annual species, field establishment suggested.
Field establishment: readily germinates and establishes in areas with low salinity.
Special considerations: prefers low salinity soils, found in areas receiving freshwater inflows.
Found in San Diego County at SBA, SWM, FS, SDL, SEL, BL, SMR

Lasthenia glabrata coulteri L. ssp. ***coulteri*** Gray (Asteraceae). **Coulter's goldfields.** FACW (although our observations here suggest this should be OBL)
Native annual forb; erect branching stem; 5 to 15 cm tall.
Flowers: yellow composite flowers, 1 to 2 cm diameter; insect pollinated.
Elevational range: 6.8 to 10.0 feet (median 7.7) MLLW; 1.3 to 2.3 m NGVD.
Phenology
Germination: Nov–Apr, depending upon rainfall and soil moisture.
Shoot production: germination flowering.
Flowering: Feb–May.
Seed collection: Mar–May/Jun.
Restoration notes
Seed collection, processing, and storage: seeds should be collected shortly after senescence, dried, then stored at cool temperatures.
Propagation techniques: seeds germinate readily in moist, low salinity soils.
Field establishment: naturally germinates with high soil moisture associated with winter rains, but not until flooded sites drain. Establishes best in areas with lowered salinity.
Special considerations: threatened species with showy yellow flowers. It does well in depressions or along salt panne margins that collect fresh or low salinity water.
Found in San Diego County at TJE, SWM, LPL, SEL

Limonium californicum (Boise) Heller (Plumbaginaceae). **Sea lavender.** OBL
Native perennial forb; rosettes with oblanceolate leaves to 30 cm, inflorescences to 75 cm high.
Flowers: small lavender flowers on inflorescence spikelets; insect pollinated.
Reproduction: primarily sexual, with occasional short lateral subsurface branching.
Elevational range: 4.5 to 7.6 feet (median 6.0) MLLW; 0.6 to 1.54 m NGVD.
Phenology
Germination: Dec–May.
Leaf production: Feb–Oct.
Flowering: Apr–Sep.
Seed collection: Sep–Nov (best Oct).
Restoration notes
Seed collection, processing, and storage: seeds can be harvested by collecting whole inflorescence stems, then shaking out flowers with seeds, or stripping them from the panicle. Seeds remain in the flower until they senesce. Seeds/flowers should be dried, then stored at cool temperatures. Seeds best if used within 1 year.
Propagation techniques: propagates readily from seed or plugs (larger plants will require more of the deep taproot).

Field establishment: seeds are highly germinable, establishing in open areas in wetter years. Seedlings, plugs, or nursery plants establish readily in restoration sites with sufficient soil moisture. Plants do better in well-drained soil, may require irrigation early in establishment phase at higher elevations.

Special considerations: seed is eaten by insect herbivores while still on the inflorescence, so should be collected soon after maturation.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL

Lycium californicum Nutt. (Solanaceae). **Boxthorn.** NLR

Native perennial shrub; densely branched woody stems with small (<20 mm) succulent awl-like leaves; 20 to 150 cm tall, 50 to 200 cm wide.

Flowers: perfect, lavender-white, 5 to 10 mm diameter; insect pollinated.

Reproduction: sexual and asexual, forming rhizomatous clones.

Elevation range: 7.0+ feet MLLW; 1.35+ m NGVD.

Phenology

Germination: not observed, presumably with winter rains.

Shoot/leaf production: with onset of winter rains; drought deciduous; new ramets produced over winter.

Flowering: approximately 1 month after first rains. Flowering may occur again after 3 to 4 months in years where early rains continue throughout the spring.

Seed collection: 2 to 3 weeks after flowering (usually Jan–Feb).

Restoration notes (see James 1999 for additional information)

Seed collection, processing, and storage: seeds (in berries) are best collected within 2 weeks of setting, after which they become rarer due to frugivorous birds. Berries must be picked by hand (2 seeds/berry) and then stored in a cool place. Berries will begin to mold after about 1 week at room temperature or 3 to 4 weeks at 40°F. Seeds should be separated from berries before molding occurs, dried and then stored in a cool place. Separated seeds can be stored for at least a year.

Propagation techniques: cuttings root easily and flower quickly. Cuttings may require initial irrigation after introduction to a restoration site. Cuttings can be collected at any time of year and then, after dipping freshly cut stems in an aqueous solution of rooting hormone, should be planted in a 2:1 vermiculite:perlite mixture and watered daily. Seeds should be soaked under running water for at least 12 hours and then transferred directly to moist soil. Germination rates are generally low, from 5 to 10% in the greenhouse.

Field establishment: naturally establishes from new shoots following rains. Can be introduced to a restoration site as cuttings, seedlings, or salvaged adults.

Special considerations: a species mostly restricted to salt marsh transition areas, tolerant to tidal inundation, although seedlings need low salinity gaps to germinate. Seedlings seem to be more drought tolerant than cuttings, and may not require irrigation. This is potentially useful in restoration, as irrigation will encourage exotic invasions. *L. californicum* occasionally occurs on coastal bluffs, but collection of seeds or cuttings from these populations should not be used for wetland restoration.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, KFR[2], LPL, SMR

Monanthochloe littoralis Engelm.(Poaceae). **Shoregrass.** OBL

Native perennial grass; prostrate stems forming dense clonal domes 10 to 35 cm high; may also produce stolons that sparsely interdigitate with other species.

Flowers: inconspicuous spikelets (to 1 cm) nearly concealed in leaves, female pistils protruding; wind pollinated.

Reproduction: primarily asexual, rooting at the nodes of stolons; dioecious.

Elevational range: 5.0 to 10.2 feet (median 6.9) MLLW; 0.74 to 2.3 m NGVD.

Phenology

Germination: Feb–May (observations few — rainfall dependent?).

Shoot/leaf production: Feb–Oct.

Flowering: Mar–Jul.

Seed collection: Jun–Sep.

Restoration notes

Seed collection, processing, and storage: seeds are small and difficult to collect. They should be collected over the summer, air dried, and stored in cool temperatures.

Propagation techniques: from seed or cuttings. Stolon cuttings root well in moist soil. Stolons should be layered horizontally just below the soil surface, with leafy shoots at each node projecting above ground.

Field establishment: clones establish new growth with wide ranging stolons. Seedlings can be found in the field, but are rare in mature communities. In restoration sites, establishes well from plugs, rooted cuttings, or nursery plants.

Special considerations: an important high marsh species, and the preferred host to the federally listed endangered species *Cordylanthus maritimus* ssp. *maritimus* (Fellows 1999).

Found in San Diego County at TJE, ORE, SBA, SWM, FS, KFR, LPL, SDL, SEL, SMR

Ruppia maritima L. (Potamogetonaceae). **Ditch-grass.** OBL

Native submersed perennial forb; multi-branched, thread-like stems and leaves arise from the nodes of rhizomes.

Flowers: minute inflorescences enclosed in the sheath at the base of leaves, two flowers per inflorescence; each flower produces 4 fruits that extend individually to the surface on 2 to 25 mm peduncles (Hurley 1990).

Reproduction: sexual and asexual, with vegetative establishment from new shoots arising at the nodes of rhizomes.

Elevation range: variable, submersed in near shallow waters.

Phenology

Phenology has not been studied in San Diego, although plants have been observed there throughout the summer months. In Vancouver, BC, Harrison (1982) found seeds of the annual population germinated in April, with biomass peaking between June and July, and up to 95% of all shoots reproductive at that time.

Restoration notes

Considerations: an aquatic species growing subtidally or in channels in San Diego, and intertidally as one proceeds further north. It is an important source of food for marsh birds and waterfowl. Some consider this a weedy species and pest; may compete with *Zostera marina* in south San Diego Bay.

Found in San Diego County at SEL, SDB, LPL

Salicornia bigelovii Torrey (Chenopodiaceae). **Annual pickleweed.** OBL.

Native annual forb; erect succulent jointed stems without leaves, 10 to 40 cm tall; mature plants branching from the midpoint up.

Flowers: green inflorescences at branch tips, swelling and turning yellow-green late in the season as the rest of the plant senesces; selfing and outcrossing, wind-pollinated.

Elevation range: 4.5 to 7.2 feet (median 5.2); MLLW; 0.6 to 1.4 m NGVD.

Phenology

Germination: Nov–Mar, dependent upon local temperature and rainfall patterns.

Shoot production: germination–Oct.

Flowering: Jun–Oct.

Seed collection: Sep–Nov (differs widely among populations and sites).

Restoration notes

Seed collection, processing, and storage: inflorescences should be collected and air-dried. Seeds strip easily from inflorescences after drying, and should be stored in cool temperatures. Viability is reduced after 1 year.

Propagation techniques: Seeds germinate readily in moist soil, with germination enhanced by low salinity.

Field establishment: natural germination is very high, with a dense seed shadow often found near the parent plants. Seeds sown directly onto the marsh surface may germinate well, but may not stay put: dry seed floats. Planting seedlings works very well, and survivorship is generally very high.

Special considerations: a prolific early colonizer of open habitat if there is a local seed source, it may eventually be replaced by or coexist with perennial marsh plain species.

Found in San Diego County at TJE, ORE, SBA, SWM, SDR, KFR

Salicornia europaea L. (Chenopodiaceae). OBL

Native annual forb; erect succulent jointed stems, densely branched, without leaves, 10 to 20 cm tall; green or red.

Flowers: green or red inflorescences at branch tips; wind-pollinated.

Elevational range: 5.1 to 9.5 feet (median 7.0) MLLW; 0.8 to 2.1 m NGVD.

Phenology

Germination: Feb–Apr, dependent upon local temperature and hydrology.

Shoot/leaf production: germination–Jun.

Flowering: May–Jul.

Seed collection: Jul–Sep (may differ widely among populations or sites).

Restoration notes

Seed collection, processing, and storage: inflorescences should be collected and dried. Seeds strip easily from inflorescences after drying, and should be stored in cool temperatures.

Propagation techniques: seeds germinate readily in moist soil, with germination enhanced by lowered salinity.

Field establishment: natural germination is relatively high, with a seed shadow often found near the parent plants. Seeds sown directly into restoration sites may germinate well, but may not stay put: dry seed floats. Seedlings establish readily, and survivorship is generally high.

Special considerations: a relatively uncommon species in San Diego County salt marshes, probably due to restrictive habitat requirements: found primarily in relatively open salt pannes subject to periodic freshwater runoff.

Found in San Diego County at SWM, LPL, SEL, BL, SMR

Salicornia subterminalis Parrish; also *Arthrocnemum subterminale* Ferren (Chenopodiaceae). **Glasswort.** OBL

Native perennial subshrub; erect succulent new growth from decumbent woody stems; 10 to 50 cm tall.

Flowers: hermaphroditic; inflorescences develop apically, with subsequent growth rendering them subterminal; wind-pollinated.

Reproduction: primarily asexual from new shoots arising at the nodes of rhizomes.

Elevational range: 5.8 to 10.3 feet (median 7.2) MLLW; 0.99 to 2.36 m NGVD.

Phenology

Germination: Dec–Mar, depending upon temperature and rainfall.

Shoot/leaf production: Feb–Aug.

Flowering: Mar–Oct.

Seed collection: Oct–Dec (best Nov).

Restoration notes

Seed collection, processing, and storage: collect inflorescences and air dry, seeds fall out or may remain in fragments; store in cool temperatures.

Propagation techniques: seed is highly germinable and promoted by lowered salinity. Cuttings will root with care, but not well.

Field establishment: plants establish naturally from new shoots arising at the nodes of rhizomes, with seedling establishment probably rare in mature marsh communities. In restoration sites, establishes very well from seedlings, also from plugs, rooted cuttings, or nursery plants.

Special considerations: an important dominant, but slow growing clonal species of the high marsh; its presence is characteristic of high marsh habitat.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, SMR

Salicornia virginica L.; also *Sarcocornia virginica* Ferren (Chenopodiaceae). **Pickleweed**. OBL

Native perennial forb; erect succulent herbaceous stems without leaves from decumbent/ascending woody growth; 20 to 100+ cm tall; highly variable.

Flowers: sequentially hermaphroditic, female flowers mature first, with some overlap; apical inflorescences resemble stems with shortened internodes; wind-pollinated.

Reproduction: readily from seed, plus asexually from new shoots rooting at the nodes of decumbent stems, and from fragments.

Elevation range: 4.2 to 9.2 feet (median 5.4) MLLW; 0.5 to 2.0 m NGVD.

Phenology

Germination: Nov–Jun, depending upon temperature and soil moisture.

Shoot/leaf production: Feb–Sep.

Flowering: Jul–Dec.

Seed collection: Oct-Dec (best Nov–early Dec).

Restoration notes

Seed collection, processing, and storage: collect and air-dry inflorescences, seeds fall out or may remain in fragments; store at cool temperatures. Seed stored at PERL has remained viable for 2+ years.

Propagation techniques: propagated from seed, cuttings, or plugs. Cuttings root well in sandy soil or water. Seed is highly germinable, enhanced by lowered salinity.

Field establishment: a very weedy species, establishing naturally from seed or vegetatively from fragments. in restoration sites, establishes very well from seedlings, plugs, rooted cuttings, or nursery plants. Cuttings may be rooted directly in the ground at low to mid intertidal elevations.

Special considerations: grows aggressively and may outcompete other slower growing species. It may be best to introduce *S. virginica* in fewer numbers as it can

dominate an open site after 1 year from abundant seedling establishment followed by abundant growth. It provides critical nesting habitat for the state listed Belding's Savannah sparrow.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, SLR, SMR

Spartina foliosa Trin. (Poaceae). **Cordgrass.** OBL

Native perennial grass; erect culms from a creeping rhizome; 35 to 130 cm tall, inflorescence stems to 150 cm.

Flowers: green inflorescences; wind-pollinated.

Reproduction: primarily asexual from new shoots produced at the nodes of rhizomes and base of culms. Clones produce many flowering shoots, but mature seed production is generally low and few seedlings are observed. Although seedling establishment is rarely high, it varies greatly among sites and years (Ward 2000).

Elevational range: 4.1 to 6.3 feet (median 5.0) MLLW; 0.47 to 1.14 m NGVD.

Phenology

Germination: Feb–Apr (often restricted to periods of freshwater inflow and lowered salinity).

Shoot production: Feb–Aug.

Flowering: Jul–Oct.

Seed collection: Sep–Nov.

Restoration notes

Seed collection, processing, and storage: seed should be refrigerated dry for 2 to 4 weeks and refrigerated in the dark in salt or fresh water (results of experimental trials comparing salt vs. freshwater storage are not consistent). Seed viability decreases after 4 months in storage (Trnka 1998).

Propagation techniques: plants propagate well from 3 to 6 inch plugs with rooted shoots. Larger plugs or blocks of sod may be subdivided just prior to planting. Viable seeds do not germinate well (<10% in tests) and both greenhouse and field establishment is difficult.

Field establishment: natural establishment from seedlings is generally rare and is likely enhanced by low salinity gaps during heavy rainfall years. Once established, a clone will spread out to colonize available space, expanding up to 1.5 m (radius) per year by the second year. In restoration sites, plugs establish very well, but seedling mortality is high.

Special considerations: seed set is variable and many are nonviable, and much of the viable seed may be lost to insect granivores. Seeds should be collected at shattering, as mature seed is shed. Multiple harvests may increase the probability of collecting good seed prior to dispersal or herbivory loss.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, SDL, BL

Spergularia macrotheca (Hornem.) Heynh. (Caryophyllaceae). **Beach sand-spurry.** FAC+

Native perennial forb; prostrate to ascending stems; 5 to 30 cm tall.

Flowers: purple to pink flowers, five petals, 7 to 9 mm diameter, with 9 to 10 stamens.

Reproduction: sexual.

Elevation range: 6.7+ feet MLLW; 1.25+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.

Shoot production: Nov–Jul.

Flowering: Feb–Jul.
Seed collection: Mar–Jul.

Restoration notes

Seed collection, processing, and storage: tiny seeds can be collected by shaking mature flowers and then dried.
Propagation techniques: seeds germinate readily in moist, low salinity soils.
Field establishment: little is known, thought to be similar to *S. marina*.
Special considerations: rare in the salt marsh, it is found near dunes and areas receiving freshwater influence.

Found in San Diego County at TJE, SWM

Spergularia marina (L.) Griseb. (Caryophyllaceae). **Salt marsh sand-spurrey.** OBL

Native annual-facultative perennial forb; prostate to ascending stems; 5 to 20 cm tall.
Flowers: purple to pink flowers, five petals, 2 to 5 mm diameter, with 2 to 5 stamens.
Reproduction: sexual.
Elevation range: 6.4+ feet MLLW; 1.2+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.
Shoot production: Nov–July.
Flowering: Feb–July.
Seed collection: Mar–July.

Restoration notes

Seed collection, processing, and storage: tiny seeds can be collected by shaking mature flowers and then dried.
Propagation techniques: seeds germinate readily in moist, low salinity soils.
Field establishment: naturally germinates and establishes in areas with lowered salinity and high moisture.
Special considerations: a facultatively perennial species, widespread but sparse in transitional habitats. It often occurs in areas receiving freshwater influence.

Found in San Diego County at TJE, SBA, SWM, FS, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Suaeda calceoliformis (Hook.) Moquin (Chenopodiaceae). **Horned sea-blite.** FACW+

Native annual forb, occurring in two ecomorphs:
transitional zone <u>ecomorph</u> — ascending stems to 50 cm; with tightly ascending leaves <40 mm.
salt marsh <u>ecomorph</u> (not previously described) –- much smaller stature, with purplish-red ascending stems from 10 to 25 cm; tightly ascending leaves <30 mm (not shown in drawings).
Flowers: 1 to 4 mm round purple/maroon flowers in clusters of 3 to 5, at the base of green/red bracts smaller than the leaves. Transitional zone morph has an obviously enlarged calyx (horned/winged morphology; see drawings), while salt marsh morph *lacks* the enlarged calyx; wind-pollinated.
Elevation range:
transitional zone ecomorph 7.2+ ft MLLW; 1.4+ m NGVD;
salt marsh ecomorph 5.5 to 8.8 feet (median 6.8) MLLW; 0.9 to 1.9 m NGVD.

Phenology of the salt marsh ecomorph

Germination: Dec–Apr (dependent upon soil moisture and salinity).
Leaf production: germination–Jul.

Flowering: May–Sep.
Seed collection: Jun–Oct.

Restoration notes

Seed collection, processing, and storage: seeds can be harvested by collecting whole inflorescences, then stripping mature flowers with seed from the inflorescences. Seeds will remain in the flower until they senesce. Seeds should be air-dried and stored at 5°C.

Propagation techniques: plants grow well from seedlings.

Field establishment: establishes naturally from seed. In restoration sites, seedlings should be planted well before the end of spring rains.

Special considerations: the salt marsh ecomorph is found in two salt marsh habitats: on the margins of pannes interspersed with salt panne vegetation, and in upper marsh plain habitat co-occurring with species such as *S. virginica*, *F. salina,* and *L. californicum*.

Found in San Diego County at TJE, SWM, SDR

Suaeda esteroa Ferren and Whitmore (Chenopodiaceae). **Sea blite.** OBL

Native short-lived perennial subshrub; upright succulent to 100 cm; with overlapping, curved lanceolate leaves 20 to 40 mm.

Flowers: 1 to 2 mm round green flowers in clusters in the apices of apical leaves.

Reproduction: sexual, flowers are wind-pollinated.

Elevational range: 4.6 to 7.0 feet (median 6.0) MLLW; 0.62 to 1.35 m NGVD.

Phenology

Germination: Jan–May.
Leaf production: Feb–Oct.
Flowering: Jul–Dec.
Seed collection: Oct–Dec (best Nov/early Dec).

Restoration notes

Seed collection, processing, and storage: seeds can be harvested by collecting whole inflorescence or stripping flowers with seeds from the inflorescence. Seeds remain in the flower until they senesce. Seeds should be dried, then stored at 5°C. Seed germinability remains high after 2 years.

Propagation techniques: plants grow well from seedlings or tip cuttings rooted in sandy soil.

Field establishment: establishes naturally from seed. In restoration sites, seedlings establish well, and may flower after one season.

Special considerations: a short-lived species, it may flower in year one, or small juveniles may persist in the understory waiting for a canopy opening. Based on field observations from TJE and greenhouse studies, adults rarely flower more than twice.

Found in San Diego County at TJE, ORE, SBA, SWM, KFR

Suaeda moquinii (Torrey) Greene (Chenopodiaceae). **Bush seepweed.** FAC+

Native perennial subshrub; upright succulent to 120 cm; with overlapping lanceolate leaves 20 to 40 mm; inflorescence stems to 2 mm, with nonoverlapping bracts smaller than leaves.

Flowers: 1 to 2 mm round green flowers in clusters of 1 to 12 in the apices of inflorescence bracts; ovary pear-shaped, seed to 1 mm.

Reproduction: sexual, flowers are wind-pollinated.

Elevation range: 7.8+ feet MLLW; 1.6+ m NGVD.

Phenology

Germination: not observed.

Leaf production: Feb–Oct.

Flowering: Apr–Jul.

Restoration notes

Seed collection, processing, and storage: seeds can be harvested by collecting whole inflorescence or stripping flowers with seeds from the inflorescence. Seeds remain in the flower until they senesce. Seeds should be dried, then stored at cool temperatures.

Propagation techniques: plants grow well from seedlings or tip cuttings rooted in sandy soil.

Field establishment: establishes naturally from seed.

Special considerations: an upland transitional species co-occurring with *S. subterminalis* and *Lycium californicum*.

Found in San Diego County at ORE, SWM

Suaeda taxifolia Standley (Chenopodiaceae). **Wooly sea-blite.** FACW+

Native a perennial shrub; upright, branching shrub to 150 cm; with succulent, overlapping lanceolate leaves, bracts <30 mm; ovary pear-shaped, with obvious neck (*S. californica* <80 cm; ovary conic, without a neck; leaves to 35 mm).

Flowers: radial green flowers, 1 to 3 mm dia., in clusters of 1 to 3; bisexual, or laterally pistillate; calyx hairy (*S. californica* flowers 2 to 3 mm dia., in clusters of 1 to 5).

Reproduction: sexual.

Elevation range: 6.8+ feet MLLW; 1.3+ m NGVD.

Phenology

Germination: not observed.

Shoot/leaf production: Feb–Aug.

Flowering: Apr–Jun.

Seed collection: Jun–Jul.

Restoration notes

Seed collection, processing, and storage: collect seed by stripping flowers with seed from inflorescences, store in cool temperatures.

Propagation techniques: propagates well from cuttings in moist sandy soil. Seeds germinate readily with freshwater.

Field establishment: from seed.

Special considerations: a medium to relatively large shrub found on the margins of salt marshes, provides important wildlife habitat.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, KFR, LPL, AHL

Triglochin concinna Burtt Davy (Juncaginaceae). **Arrow-grass.** OBL

Native perennial graminoid; cespitose, with cylindrical succulent leaves from rhizomes, 2 mm diameter basally, 10 to 25 cm tall; inflorescences to 35 cm.

Flowers: 2 to 3 mm diameter flowers with white/purple hairs, along an inflorescence spike.

Reproduction: primarily asexual from tillers.

Elevation range: 5.2 to 6.8 feet (median 5.8) MLLW; 0.8 to 1.3 m NGVD.

Phenology

Germination: Dec–Mar (influenced by rainfall patterns).

Leaf production: Fall–June (remains fairly dormant until Jan/Feb).

Flowering: inflorescences develop over 3 to 4 weeks from Jan–Jun.
Seed collection: Apr–Jun (best May/Jun).

Restoration notes

Seed collection, processing, and storage: seeds should be collected shortly after inflorescences turn brown, dried and stored in cool temperatures. Seeds last up to 3+ years.

Propagation techniques: plants propagate well from seed or plugs collected from established turf.

Field establishment: naturally establishes from tillers, with seedling establishment probably rare in mature communities. Seedlings or plugs generally do well in restoration sites if established early (late fall). New plants compete poorly in the summer months. Prefers moist, fertile soil; establishes well where other species shade the soil surface.

Special considerations: a winter active perennial, the shoots die back by mid summer, with new shoots establishing in the fall, then remaining dormant until Jan–Feb.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR

Zostera marina L. (Zosteraceae). **Eelgrass.** OBL

Native perennial or annual marine graminoid; a creeping rhizome, 2 to 5 mm dia. giving rise to a short nonreproductive stem, or an elongated and branched reproductive shoot to 3 m long; 1 to 2 shoots per node; a variable number of leaves per shoot, to 160 cm long and 12 mm wide. Annual form is less common, much less robust with shorter, narrower leaves.

Flowers: inflorescences flattened, spadices 3 to 6 cm long, spathe 4 to 10 cm; pistillate and staminate flowers alternating in two rows; hydrophilous pollination in the water column. Multiple fruits produced on each inflorescence, with one seed per fruit.

Reproduction: from seed and asexually from new shoots produced at the nodes of rhizomes. Perennial ramets commonly flower in their second year. Recruitment dynamics from seed are not well understood and may vary widely among populations.

Elevational range: variable, ranging from subtidal to low intertidal (annual form to mid-intertidal); strongly dependent upon water clarity (lower limit) and abiotic factors such as desiccation stress or disturbance (upper limit); locally from –8.0 to 0.0 feet MLLW (rarely intertidal); –3.2 to –0.8 m NGVD (–10 m to 1 m in Pacific Northwest; sub- and intertidal).

Phenology

Germination: Oct–Apr (influenced by water temperature).
Shoot production: Feb–Sep.
Flowering: May–Jun.
Seed collection: Sep–Nov.

Restoration notes

Seed collection, processing, and storage: seeds collected from intact mature inflorescences with mature darker fruits in evidence; inflorescences mature asynchronously-apically along the reproductive shoot. Reproductive shoots may be collected and held in a tank (or tanks) with flowing seawater until seeds dehisce. For storage, seeds should be separated from decaying tissue, stored in cold flowing seawater or under refrigeration. See Fonseca et al. (1998) for guidelines on seagrass restoration techniques, and Orth et al. (1994) for a discussion of issues in *Zostera* restoration from seedlings.

Propagation techniques: we have not tested germination techniques. Others have found that germination ecology varies with geographical location. Seeds germinate

asynchronously, peaking after water temperature begins to rise after the winter lows. Germination from perennial plants on the Chesapeake Bay was highest around 15°C under anoxic conditions, similar to that experienced by buried seeds under natural conditions (Moore et al. 1993). Others have suggested germination may be enhanced by lowered salinity, around 15 ppt (Orth and Moore 1983).

Field establishment: natural establishment from seedlings in established perennial beds may be rare. Annual populations germinate readily in the field. Establishment from perennial populations is lower and variable. Orth et al. (1994) found establishment of seedlings from hand-dispersed seeds ranged from 4 to 40%, with little post-sowing dispersal. Once established, a clone will spread out to colonize available space under favorable conditions. In restoration sites, plugs anchored with sediment remaining around intact roots establish well, expanding in year 2. Bare root transplants establish poorly. To maximize spread rates, transplants should be arrayed in many small patches rather than few large patches (Olesen and Sand-Jensen 1994). Establishment is strongly dependent upon the care exercised in handling fragile transplants and will also vary with site quality.

Special considerations: seed or transplants should be collected from populations with relatively high genetic diversity — low diversity restorations may perform poorly (Williams and Davis 1998).

Found in San Diego County at TJE, SDB, SDR, intertidally at MB

Nonnative species

Atriplex semibaccata R. Br. (Chenopodiaceae). **Australian saltbush.** FAC

Nonnative perennial forb-subshrub; ascending stems, to 40 cm, spreading to >3 m wide; leaves variable, oblong to elliptic 8 to 35 mm.

Flowers: female flowers in axils of leaves, male inflorescence terminal.

Reproduction: sexual; monoecious.

Elevation range: 7.2+ feet MLLW; 1.4+ m NGVD.

Phenology has not been studied closely here. Active growth from late spring through fall, flowering through late summer.

Management notes

The only nonnative perennial species in southern California salt marshes that is tolerant of higher salinity. May form locally dense clonal mats. Very widespread.

Found in San Diego County at TJE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR

Bassia hyssopifolia (Pallas) Kuntze (Chenopodiaceae). **Hyssop-leaved bassia.** FAC

Nonnative annual forb; erect stems with ascending to erect branches, 10 to 40 cm.

Flowers: 1 to 2 mm, calyx densely wooly with hooked spines.

Elevation range: 6.8+ feet MLLW; 1.3+ m NGVD.

Phenology has not been studied here.

Management notes

Found in drier upland-transition areas, establishes well in disturbed areas during windows of low salinity associated with seasonal rains.

Found in San Diego County at TJE, ORE, SWM, FS, KFR

Cotula coronopifolia L. (Asteraceae). **Brass-buttons.** FACW+

Nonnative annual forb; erect or ascending forb; 5 to 30 cm tall.

Flowers: yellow composite with 2 to 10 cm disk flowers, no ray flowers; insect pollinated.

Elevation range: 5.5+ feet MLLW; 0.9+ m NGVD.

Phenology

Germination: Nov–Mar, following winter rainfall or flooding.

Shoot production: germination–Apr.

Flowering: Mar–Apr.

Seed dispersal: Mar–Jul.

Management notes

Opportunistic species found at a range of elevations experiencing low or lowered salinity. Restricted to brackish conditions, dense monocultures occur in habitats experiencing freshwater runoff.

Found in San Diego County at TJE, SWM, SDR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Limonium ramosissimum (Poir.) Maire **ssp. *provinciale*** Pignatti (Plumbaginaceae). NLR

Nonnative perennial forb; dense rosette of lanceolate leaves to 15 cm, inflorescences to 40 cm high.

Flowers: small blue flowers on inflorescence spikelets; insect-pollinated.

Reproduction: sexual, with short lateral subsurface branching giving rise to new rosettes.

Elevation range: approx. 5.2 + feet MLLW; 0.8 + m NGVD.

Phenology

Germination: not observed here.

Leaf production: Feb–Oct.

Flowering: Apr–Jul.

Seed dispersal: May–Jul (perhaps longer — based on few observations).

Management notes

This subspecies of *L. ramosissimum* may be distinguished from *L. californicum* as a dense rosette of shorter and narrower leaves, and by a shorter inflorescence with flowers more densely grouped along the spikelets. It is an ornamental escape with the potential to aggressively displace native species. It has been found in marsh plain habitat co-occurring with *S. virginica*, *J. carnosa*, and *D. spicata* at Agua Hedionda Lagoon, and in the high marsh transition zone at San Elijo Lagoon in San Diego County. A different subspecies of *L. ramosissimum* has been found at Carpinteria Salt Marsh, California, where it has aggressively displaced other marsh plain species including the federally listed *Cordylanthus maritimus* ssp. *maritimus*, while resisting intense efforts at eradication (Hubbard and Page 1997).

Found in San Diego County at SEL, AHL

Lolium multiflorum Lam. (Poaceae). **Italian ryegrass.** NLR

Nonnative annual grass; erect culms; 20 to 50 cm.

Flowers: green inflorescences, spikelets alternating in plane along the axis; wind pollinated.

Elevation range: 7.8+ feet MLLW; 1.6+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall patterns.

Shoot production: germination–Jun.

Flowering: Mar–Jun.

Seed dispersal: All summer.

Management notes

An aggressive species, it can be found densely in moist but nonsaline wetland-upland transitions, a good indicator of the saline habitat margin.

Found in San Diego County at TJE, ORE, SBA, FS, LPL, SEL, BL, AHL, BVL, SMR

Lythrum hyssopifolium L. (Lythraceae). **Hyssop loosestrife.** FACW

Nonnative annual forb; decumbent stems; 2 to 7 cm tall, spreading to 40 cm wide.

Flowers: purple to pink flowers, 3 to 6 mm diameter; insect pollinated.

Elevation range: 6.0+ feet MLLW; 1.05+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.

Shoot production: germination–Jun.

Flowering: Apr–Jun.

Seed dispersal: May–Jul.

Management notes

Low tolerance of salinity. Not widely distributed, found primarily in areas with freshwater runoff.

Found in San Diego County at LPL, SDL, SEL, AHL, SMR

Mesembryanthemum crystallinum L. (Aizoaceae). **Crystalline iceplant.** FACU

Nonnative annual forb; dense ground cover, with prostrate to ascending stems; 10 to 30 cm tall.

Flowers: white composite-like flowers 8 to 10 mm diameter, with 5 sepals; insect pollinated.

Elevational range: 7.8+ feet MLLW; 1.6+ m NGVD.

Phenology

Germination: asynchronous, beginning in the fall with the first seasonal rains. Subsequent germinations take place throughout the year (Vivrette and Muller 1977).

Shoot production: germination–Aug.

Flowering: Mar–Aug.

Seed dispersal: from the first rains or heavy fog through the fall and winter (seeds released as dried capsules open in response to moisture (Vivrette and Muller 1977).

Management notes

Common exotic species found at the upland-wetland transition. Less tolerant of salinity than *M. nodiflorum*, this broad-leafed species occurs singly or as thick ground cover excluding native species.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, BL, AHL, BVL, SMR

Mesembryanthemum nodiflorum L. (Aizoaceae). **Slender-leaved iceplant.** FACU

Nonnative annual forb; dense ground cover, with prostrate to ascending stems; 5 to 20 cm.

Flowers: white composite-like flowers 4 to 5 mm diameter, with 5 sepals; insect pollinated.

Elevational range: 6.8+ feet MLLW; 1.3+ m NGVD.

Phenology

Germination: Nov–Mar, dependent upon seasonal rains.

Shoot production: germination–Jul.

Flowering: May–Jul.

Seed dispersal: summer and fall.

Management notes

Common exotic species in disturbed and naturally open areas of high salt marsh and upland-wetland transition. Highly tolerant of salinity, this species forms a broad mat of ground cover that may preclude establishment by native species.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, BL, AHL, BVL, SMR

Parapholis incurva Hubb. (Poaceae). **Sicklegrass.** OBL

Nonnative annual grass; erect-ascending culms; 5 to 20 cm tall.

Flowers: inflorescences enclosed in curving stem; wind or self-pollinated.

Elevational range: 6.4+ feet MLLW; 1.2+ m NGVD.

Phenology

Germination: Nov–Mar, in moist soil following seasonal rainfall.

Shoot production: germination–May.

Flowering: Mar–May.

Seed dispersal: all summer.

Management notes

Common and abundant exotic species in the high salt marsh, co-occurring with *S. subterminalis*, *M. littoralis*, and *C. m. maritimus*. May be extremely aggressive, with dense monocultures displacing native species.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SMR

Polypogon monspeliensis (L.) Desf. (Poaceae). **Rabbit's-foot grass.** FACW+

Nonnative annual grass; erect culms; 5 to 40 cm; inflorescences to 70 cm.

Flowers: green plume-like inflorescences; wind pollinated.

Elevation range: 6.0+ feet MLLW; 1.05+ m NGVD.

Phenology

Germination: Nov–Mar, with moist soil conditions.

Shoot production: germination–Jul.

Flowering: Mar–Jul.

Seed dispersal: all summer.

Management notes

An extremely aggressive species in areas with freshwater influence; outcompetes native high marsh halophytes where runoff or stream inflows lower salinity.

Found in San Diego County at TJE, ORE, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Rumex crispus L. (Polygonaceae). **Curlydock**. FACW-

Nonnative perennial forb; basal and cauline leaves, reproductive shoots to 120 cm.

Flowers: green flowers 1 to 3 mm, in clusters along an erect panicle inflorescence.

Reproduction: sexual.

Elevation range: 6.8+ feet MLLW; 1.3+ m NGVD.

Phenology

Germination: germination requires a window of very low salinity, although seedlings that establish may tolerate moderate increases (Zedler and Beare 1986).

Shoot/leaf/flower production: growth is primarily early spring through midsummer, as apical shoots convert to seed production.

Seed dispersal: summer–fall.

Management notes

Found in transitional zones between salt marsh and freshwater marsh, or uplands experiencing significant seasonal freshwater influence. Overwinters as a basal rosette.

Found in San Diego County at TJE, SWM, FS, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Sonchus asper L. (Asteraceae). **Prickly sow thistle.** FAC

Nonnative annual forb; erect stems; 20 to 120 cm tall; basal lobes of leaves rounded, fruits 3 ribbed and smooth; stems and leaves more robust, spines tougher than *S. oleraceus.*

Flowers: yellow composite flowers, 1 to 2 cm diameter; insect pollinated.

Elevational range: 7.0+ feet MLLW; 1.35+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.

Shoot production: germination–Apr.

Flowering: Feb–Apr.

Seed dispersal: Mar–Jun.

Management notes

Found in salt marsh transition zones in areas with seasonally depressed salinity.

Found in San Diego County at TJE, SWM, LPL, SDL, SEL, BL, AHL, SMR

Sonchus oleraceus L. (Asteraceae). **Common sow thistle.** NLR

Nonnative annual forb; erect stems; 10 to 125 cm tall; basal lobes of leaves acute, fruits 2 to 4 ribbed and wrinkled; stem and leaf spines softer than *S. asper.*

Flowers: yellow composite flowers, 1 to 2 cm diameter; insect pollinated.

Elevation range: 6.5+ feet MLLW; 1.2+ m NGVD.

Phenology

Germination: Nov–Mar, depending upon rainfall.

Shoot production: germination–Apr.

Flowering: Feb–Apr.

Seed dispersal: Mar–Jun.

Management notes

Found in salt marsh–freshwater marsh transitions.

Found in San Diego County at TJE, ORE, SBA, SWM, FS, SDR, KFR, LPL, SDL, SEL, BL, AHL, BVL, SLR, SMR

Table A2.1 U.S. Fish and Wildlife Service wetland species ratings, according to the probability that a species will be found in a wetland habitat. In the text, codes followed by a + indicate nearer the higher probability in each range, while a – indicates nearer the lower probability (e.g., FAC+ means that this species occurs in wetlands nearly 66% of the time).

USFWS species rating	Code	Probability found in wetland
Obligate Wetland	OBL	>99%
Facultative Wetland	FACW	67–99%
Facultative	FAC	34–66%
Facultative Upland	FACU	1–33%
Obligate Upland	UPL	<1%
Not Listed for this Region	NLR	—

1. U.S. Fish and Wildlife Service wetland indicator categories.
2. Introduced at that site.
3. + indicates nearer the higher probability in Table A2.1 — indicates nearer the lower probability in Table A2.1.

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Appendix 3

Distribution of plant species in coastal wetlands of San Diego County

Gary Sullivan & Gregory B. Noe

Native (N) and exotic (E) species distributions, with habitat type for each species. Wetlands read from south (left) to north (right). MP = marsh plain; HM = high marsh; ST = subtidal; FWT = freshwater transition; UPT = upland transition. For species found in more than one habitat, the habitats listed indicate their range of occurrence. X = observed; I = observed, but introduced. Surveyed by Sullivan and Noe April 1998. Despite exhaustive efforts, some rare species may have been missed [in particular *L. glabrata* at San Dieguito Lagoon (previously reported by L. Parsons, but site unknown) and at Santa Margarita River (access restricted)]. See Appendix 2 for salt marsh abbreviation codes.

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Distribution of plant species in coastal wetlands of San Diego County

Species	Habitat	N/E	TJE	ORE	SBA	SWM	FS	SDR	KFR	LPL	SDL	SEL	BL	AHL	BVL	SLR	SMR
Amblyopappus pusillus	HM-UPT	N	X	X	X	X				X				X	X		X
Atriplex californica	HM-UPT	N	X			X				X							
Atriplex triangularis	FWT	N	X	X		X	X	X	X	X	X	X	X	X		X	X
Atriplex watsonii	HM	N	X		X	X		X	X	X							
Batis maritima	MP	N	X	X	X	X	X	X	X								
Cordylanthus maritimus	MP-HM	N	X			X											
Cressa truxillensis	HM	N	X	X	X	X				X	X	X	X	X			
Cuscuta salina	MP	N	X		X	X	X	X	X	X	X	X	X	X			
Distichlis spicata	MP-UPT	N	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Eriogonum fasciculatum	UPT	N	X		X	X	X		I	X	X	X	X	X			
Frankenia palmeri	UPT	N	I			X			I								
Frankenia salina	MP-HM	N	X	X	X	X	X	X	X	X	X	X	X	X	X		X
Heliotropium curassivicum	UPT	N	X			X		X	X	X		X	X			X	X
Hutschinsia procumbens	HM-UPT	N				X					X						
Isocoma menziesii	UPT	N	X	X	X	X	X	X	I	X	X	X	X	X	X		X
Jaumea carnosa	MP	N	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Juncus acutus	FWT	N	X		X	X	X	X		X	X	X	X		X	X	X
Juncus bufonius	FWT	N			X	X	X				X	X	X				X
Lasthenia glabrata	HM	N	X			X				X		X					
Limonium californicum	MP-HM	N	X	X	X	X	X	X	X	X	X	X					
Lycium californicum	HM-UPT	N	X	X	X	X	X		I	X							X
Monanthochloe littoralis	MP-HM	N	X	X	X	X	X	X	X	X	X	X					X
Ruppia maritima	ST	N	X									X					
Salicornia bigelovii	MP	N	X	X	X	X		X	X								
Salicornia europaea	HM	N				X				X		X	X				X
Salicornia subterminalis	HM	N	X	X	X	X	X	X	X	X	X	X	X	X			X
Salicornia virginica	MP-HM	N	X	X	X	X	X	X	X	X	X	X	X	X			X
Spartina foliosa	MP	N	X	X	X	X	X	X	X		X		I				

Spergularia macrotheca	UPT	N				X											
Spergularia marina	HM-UPT	N	X		X	X	X		X	X	X	X	X	X	X	X	X
Suaeda calceoliformis	MP-HM	N	X			X		X				X					
Suaeda esteroa	MP-HM	N	X	X	X	X			X								
Suaeda moquinii	UPT	N	X	X		X											
Suaeda taxifolia	UPT	N	X	X	X	X	X		X	X				X			
Triglochin concinna	MP	N	X	X	X	X	X	X	X								
Zostera marina	ST	N			X	X		X	X								
Atriplex semibaccata	HM-UPT	E	X		X	X	X	X	X	X	X	X	X	X	X	X	
Bassia hyssopifolia	UPT	E	X	X		X	X		X								
Cotula coronopifolia	MP-FWT	E	X			X		X		X	X	X	X	X	X	X	
Limonium ramossimum	MP-UPT	E										X		X			
Lolium multiflorum	UPT	E	X	X	X		X			X		X	X	X	X		X
Lythrum hyssopifolium	MP-FWT	E								X	X	X		X			X
Mesembryanthemum crystallinum	UPT	E	X			X	X	X	X	X	X						
Mesembryanthemum nodiflorum	HM-UPT	E	X	X	X	X	X	X	X	X	X		X	X	X		X
Parapholis incurva	HM-UPT	E	X	X	X	X	X	X	X	X	X	X	X	X	X		X
Polypogon monspeliensis	MP-FWT	E	X	X		X	X	X	X	X	X	X	X	X	X	X	X
Rumex crispus	FWT	E	X			X	X			X	X	X	X	X	X	X	X
Sonchus asper	UPT	E	X			X				X	X	X	X	X			X
Sonchus oleraceus	UPT	E	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X

N= native

E = exotic

HM = high marsh

MP = marsh plain

ST = subtidal

FWT = freshwater transition

UPT = upland transition

Appendix 4

Habitat and elevational distribution of salt marsh plant species

Gary Sullivan

Elevation data for coastal wetland species in San Diego County (m above the 1929 mean sea level = National Geodetic Vertical Datum). Species transitioning into other habitats list only the lowest elevation. MP = marsh plain; HM = high marsh; FWT = freshwater transition; UPT = upland transition. Some elevations were estimated (est.) from their position relative to other species (from relative elevation data — benchmarks were not available).

Habitat	Species	Low	Median	Mode	High	Ref.
MP	*Spartina foliosa*	0.47	0.74	0.65	1.14	1 5 6 7 8
MP	*Salicornia bigelovii*	0.59	0.80	0.80	1.41	1 5 7 8
MP	*Batis maritima*	0.50	0.87	0.75	1.60	1 4 5 7 8
MP-HM	*Salicornia virginica*	0.50	0.87	0.90	2.02	1 4 5 7 8
MP	*Jaumea carnosa*	0.59	0.90	0.85	1.35	1 4 5 7 8
MP	*Triglochin concinna*	0.80	0.99	0.95	1.29	4 5 7 8
MP-HM	*Cuscuta salina*	0.59	1.02	1.15	1.51	1 4 5 7 8
MP-HM	*Limonium californicum*	0.59	1.05	1.15	1.54	1 4 5 7 8
MP-HM	*Suaeda esteroa*	0.62	1.05	1.00	1.35	1 4 5 7 8
MP-HM	*Frankenia salina*	0.80	1.14	1.20	1.51	1 4 5 7 8
MP-HM	*Cordylanthus maritimus*	0.76	1.29	1.05	2.20	2 4 5
MP-HM	*Suaeda calceoliformis**	0.90	1.30		1.90	5 est.
MP-HM	*Monanthochloe littoralis*	0.74	1.32	1.36	2.33	1 2 4 5 7 8
HM	*Salicornia europaea*	0.78	1.35	1.45	2.12	4 5
HM	*Salicornia subterminalis*	0.99	1.41	1.45	2.36	1 4 5 7 8
HM	*Atriplex watsonii*	1.11	1.41	1.25	2.12	4 5
HM	*Hutchinsia procumbens*	1.41	1.48		1.69	4
HM-UPT	*Amblyopappus pusillus*	1.29	1.51		1.69	4
HM	*Cressa truxillensis*	0.99	1.57		2.24	4 5
HM	*Lasthenia glabrata*	1.29	1.57		2.27	4 5
MP-UPT	*Distichlis spicata*	0.71				1 2 4 5 7
FWT	*Atriplex triangularis*	0.90				4 5 est.
MP-FWT	*Cotula coronopifolia*	0.90				4
HM-UPT	*Atriplex californica*	0.99				4 5 est.

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Habitat	Species	Low	Median	Mode	High	Ref.
MP-UPT	*Limonium ramosissimum*	1.05				4 5 est.
MP-FWT	*Lythrum hyssopifolium*	1.05				4
MP-FWT	*Polypogon monspeliensis*	1.05				4
HM-UPT	*Atriplex semibaccata*	1.11				4 5
HM-UPT	*Parapholis incurva*	1.17				4
HM-UPT	*Spergularia marina*	1.17				4 5
HM-UPT	*Mesembryanthemum nodiflorum*	1.20				4 5 est.
UPT	*Sonchus oleraceus*	1.20				4
UPT	*Spergularia macrotheca*	1.26				4
UPT	*Bassia hyssopifolia*	1.29				4
UPT	*Heliotropium curassavicum*	1.29				4
FWT	*Rumex crispus*	1.29				4 5 est.
UPT	*Suaeda taxifolia*	1.29				4 5
UPT	*Frankenia palmerii*	1.35				4 5 est.
HM-UPT	*Lycium californicum*	1.35				3 4 5
FWT	*Juncus acutus*	1.51				4
FWT	*Juncus bufonius*	1.41				4
UPT	*Eriogonum fasciculatum*	1.60				4 5
UPT	*Isocoma menziesii*	1.66				5 est.
UPT	*Lolium multiflorum*	1.60				4
UPT	*Mesembryanthemum crystallinum*	1.60				4 5 est.
UPT	*Suaeda moquinii*	1.60				5 est.
UPT	*Sonchus asper*	1.66				4 5 est.

* *S. calceoliformis* elevations are for the salt marsh ecomorph only.

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Appendix 5

Ecological and life history characteristics of common southern California salt marsh invertebrate species

Julie S. Desmond, Janelle M. West, and Gregory D. Williams

Capitella capitata
(Annelida: Polychaeta: Capitellidae)

Cosmopolitan, burrowing polychaete. Dark red or brownish, earthworm-like. Origin is unknown; it has been found in nearly every region where benthic collections have been made (Grassle and Grassle 1976). Length to 10 to 20 mm (Levin 1984*a*). Speciation of the genus *Capitella* is still unresolved (Grassle and Grassle 1976); a number of species distinguishable by gel electrophoresis are present in southern California waters (Levin 1984*a*).

Habitat

Muddy sediments in bays, estuaries. Tolerant to pollution and anoxic conditions that few other organisms can withstand (Reish 1957, 1959), it has been characterized as an indicator of polluted or unpredictable environments (Grassle and Grassle 1974).

Reproduction

In many areas, even despite great temperature fluctuations, reproduction occurs year round. Females receive spermatophores from males, which the females can store until their eggs are ripe (Porch 1970). Larvae are brooded in tubes specially constructed for reproduction; they spend only a brief time in the plankton (1 to 2 hours or up to 2 days); thus, this species shows only short-range dispersal (Levin 1984*a*). Sexual maturity is attained within one month (Morris et al. 1980).

Feeding

Subsurface-deposit feeder.

Polydora nuchalis
(Annelida: Polychaeta: Spionidae)

Tube-building polychaete worm. Length 10 to 20 mm (Levin 1984*a*).

Habitat

Mudflats and creekbeds of estuaries and bays in southern California.

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Reproduction

Eggs are deposited in strings and attached to the inside of the tube. Up to 20 egg capsules (each containing up to 230 eggs) are laid. One to eight eggs per capsule develop into larvae; the rest serve as "nurse eggs" which are consumed by the developing larvae (Wible 1984). Upon hatching (approximately 2 1/2 weeks after deposition), larvae may settle immediately. If an appropriate substrate is not available, larvae may remain in the plankton for several weeks. Generation time is about 6 months in the field (Woodwick 1955). Individuals were unable to reproduce at 15°C or lower in the laboratory, and egg capsules were found in spring, summer, and fall (but not winter) in several southern California locations (Wible 1984).

Feeding

Surface-deposit and suspension-feeder.

Streblospio benedicti

(Annelida: Polychaeta: Spionidae)

Tube-building polychaete. Native to Atlantic coast of North America, now abundant throughout the coastal U.S. and Europe (Grassle and Grassle 1974). Probably introduced to San Francisco Bay with Atlantic oysters (*Crassostrea virginica*) between the 1860s and 1910s (Carlton 1975).

Habitat

Muddy sediments; salt marshes, mudflats and creekbeds of estuaries; upper 2 to 3 cm of sediment (Levin 1984*b*). Often present in polluted areas (Reish and Winter 1954, Felice 1959) and thought of as an enrichment specialist or opportunist (Grassle and Grassle 1974).

Reproduction

Both planktotrophic (feeding, with extended periods in plankton) and lecithotrophic (nonfeeding, with limited periods in plankton) larval development have been observed in *S. benedicti* (Levin 1984*b*). Larvae are brooded for several days before release; planktotrophic larvae remain in the plankton between 10 to 21 days before settling (Levin 1984*b*). Reproduces year round, with multiple broods during its life span in California (Levin 1984*a*) and other areas. High colonization potential in disturbed or newly created areas. Responds to enrichment with increased reproductive output (Levin and Creed 1986).

Feeding

Surface-deposit and suspension-feeder; uses palps to collect food from the sediment surface or from the water column. Consumes plankton, organic aggregates, and sediments.

***Bulla gouldiana* — Bubble snail**

(Mollusca: Gastropoda: Cephalaspidea: Bullidae)

Opisthobranch snail; length to 55 mm. A large, yellow or brown mantle almost covers the shell. Ranges from Morro Bay, California, to Gulf of California, Ecuador (Morris et al. 1980).

Habitat

Found in mud and sandy mud substrates just below the mean low tide level.

Reproduction

In southern California, long, yellow strings of eggs are deposited on mud or in eelgrass during the summer; season may vary in other locations. Life span is about 1 year.

Feeding

Feeding behavior of *B. gouldiana* is not well known, but members of the family Bullidae are generally considered herbivores (Morris et al. 1980).

***Cerithidea californica* — California horn snail**

(Mollusca: Gastropoda: Mesogastropoda: Potamididae)

Extremely abundant gastropod in California salt marshes (Macdonald 1967, Race 1981, Fong et al. 1997). Ranges from central Baja California, Mexico, to Tomales Bay, California (Race 1981). Shell is black, reaching a height of 45 mm.

Habitat

Inhabits salt marsh pannes, creeks, and mudflats, and the vegetated marsh surface. There is evidence of competition with an introduced gastropod (*Ilyanassa obsoleta*) in south San Francisco Bay, which limits it to marsh panne habitats at this site (Race 1982). Individuals burrow into the substrate or migrate to the vegetated marsh surface during the winter months (Race 1981, Driscoll 1972).

Reproduction

Size at first reproduction was 22 to 24 mm in San Francisco Bay (Race 1981); snails were estimated to be approximately 2 years old at this size. Egg capsules were deposited on the sediment surface beginning in early May in San Francisco Bay (Race 1981). Eggs hatch 4 to 6 weeks after being laid (McCloy 1979). There is no extended planktonic developmental stage: settlement of veliger-stage larvae occurs immediately upon hatching (Macdonald 1967). Limited dispersal ability; adults may crawl and juveniles may float into new habitats (Race 1981).

Feeding

Surface-deposit feeder; scrapes a fluid mixture off of the sediment surface using its radula. Mainly consumes benthic diatoms, along with some detrital matter. Ceases feeding when exposed during low tide (Whitlatch and Obreski 1980).

***Macoma nasuta* — Bent-nosed clam**

(Mollusca: Bivalvia: Veneroida: Tellinidae)

Bivalve ranging from Alaska to Cabo San Lucas, Baja California Mexico. Shell is white, thin, flattened, and bent at one side, with a length to 60 mm (Morris et al. 1980).

Habitat

Common in a variety of substrates ranging from gravel to mud, 10 to 20 cm below the sediment surface in the low intertidal (Peterson 1977, Rae 1979, Morris et al. 1980). Occurs from intertidal areas to 50 m depth offshore, but it is most abundant in protected bays (Morris et al. 1980).

Reproduction

Broadcast spawner; MacGinitie (1935) noted that it spawned in the summer in Elkhorn Slough, California. Abundance did not change seasonally in Tomales Bay, California, but new recruits were most abundant in June and August (Rae 1979).

Feeding

Although some have reported *M. nasuta* to be a suspension feeder, most have classified it as a surface-deposit feeder, and recent physiological evidence (Specht and Lee 1989) supports this. The long siphons are extended to the surface of the sediment to collect microalgae and other organic material and sediments. In Carpinteria Marsh, California, surface feeding by *M. nasuta* was sufficient to create a patchy distribution of epibenthic microalgae (Page et al. 1992).

***Musculista senhousia* — Asian mussel**

(Mollusca: Bivalvia: Veneroida: Mytilidae)

Small, exotic mussel native to Asia, now introduced in Australasia, the Mediterranean, and the Pacific coast of North America (Crooks and Khim 1999). Probably transported to the U.S. along with Japanese oysters (Carlton 1975 [listed as *Musculus senhousia*]). Length to 32 mm (Crooks 1992). Shell is thin and characterized by brown zig-zags and reddish radial lines (Crooks 1992).

Habitat

Surface of intertidal and shallow subtidal soft sediments in bays and estuaries (Smith and Carlton 1975). Often associated with eelgrass or macroalgae (Reusch and Williams 1998). Typically found in brackish waters, but capable of inhabiting areas with oceanic salinity (Crooks 1992). Capable of inhabiting low-flow, soft-sediment areas that few other suspension-feeding bivalves can tolerate (Crooks 1992). When present in high densities, the mussel creates mats across sediment surface, influencing the abundance of native benthic fauna as well as physical processes (i.e., sedimentation) on the sediment surface (Crooks 1998).

Reproduction

Spawns throughout much of the year in San Diego, with a peak in summer (Crooks 1992). Spawning may be limited by temperature in other regions. Characterized by an extended planktonic stage, which may account for its high dispersal ability. Life span is approximately 2 years (Crooks 1992).

Feeding

Suspension feeder.

***Protothaca staminea* — Pacific littleneck clam**

(Mollusca: Bivalvia: Veneroida: Veneridae)

Shallow-dwelling, filter-feeding bivalve. Commercially valued in California, Oregon, and Washington (Emmett et al. 1991). Ranges from Cape San Lucas, Baja California, Mexico, to the Aleutian Islands, Alaska. Shell is relatively thick, reaches a length of about 70 mm and is characterized by radial ribs crossed by weak concentric ridges (Morris et al. 1980).

Habitat

Occupies burrows 3 to 8 cm deep in substrates ranging from mud to cobble (Quayle and Bourne 1972); prefers firm gravel or clay-gravel sediments. Occurs in bays or coves or in protected areas along the open coast (Emmett et al. 1991).

Reproduction and movements

Spawning occurs from late spring (March or April) to summer/early fall in most locations. Eggs are fertilized externally. Free-swimming larvae hatch from fertilized eggs within about 12 hours and spend about 3 weeks in the plankton, depending on water temperature (Quayle and Bourne 1972). Small clams are somewhat mobile and can use their feet to crawl to new areas (Shaw 1986); adults are sessile. Age at sexual maturity has been estimated between 1.5 and 3 years and varies with location (Paul and Feder 1976, Fraser and Smith 1928).

Feeding

Suspension feeder; uses siphons to collect particles from the water. Stable isotope evidence suggests that the clam feeds mainly on benthic algae and phytoplankton (Page 1997); detritus is also thought to be important in the diet (Chew and Ma 1987).

***Tagelus californianus* — California jackknife clam**
(Mollusca: Bivalvia: Veneroida: Psammobiidae)

Long, razor-like clam which reaches lengths of 10 to 12 cm (Morris et al. 1980). Used commercially as fish bait (Emmett et al. 1991). Ranges from Cape San Lucas, Baja California, Mexico, to Cape Blanco, Oregon (Emmett et al. 1991). Shell is thin and brownish-gray.

Habitat

Occurs in sand, mud, or muddy sand in flats and channels of protected bays and estuaries. Occupies a permanent burrow 10 to 50 cm deep, where it rapidly retreats when disturbed.

Reproduction and movements

T. californianus is a broadcast spawner; eggs are fertilized externally (Emmett et al. 1991). Little is known about the seasonality of spawning or how long larvae remain in the plankton. Maturity is reached between 60 and 120 mm total length (Merino 1981).

Feeding

Suspension feeder; occupies the upper 10 cm of its burrow while feeding (Morris et al. 1980). Feeds on phytoplankton and other suspended particles.

Grandidierella japonica
(Arthropoda: Malacostraca: Amphipoda: Corophiidae)

Exotic tube-building amphipod from Japan, first introduced to California (San Francisco Bay) in 1966 (Chapman and Dorman 1975). Now present in bays and estuaries from San Francisco to San Diego, California (Greenstein and Tiefenthaler 1997). Thought to have been introduced along with Japanese oysters (Carlton 1975), which have been imported to San Francisco Bay since 1920.

Habitat

Inhabits U-shaped tubes in intertidal and subtidal sand and mud sediments (Greenstein and Tiefenthaler 1997). Very tolerant of DDT contamination; in San Francisco Bay, its abundance was positively correlated with DDT contamination and toxicity (Ferraro and Cole 1997).

Reproduction

Eggs are incubated by the female, and juveniles are released between 7 to 10 days after the first appearance of the eggs (Greenstein and Tiefenthaler 1997). In Newport Bay, California, gravid females were collected throughout the year, although the brood size per female was lower between October and January (Greenstein and Tiefenthaler 1997). In Japan brooding females were collected between February and October (Ariyama 1996). May produce up to 12 generations per year.

Feeding

Surface-deposit or suspension feeder.

***Hippolyte californiensis* — Slender green shrimp; grass shrimp**
(Arthropoda: Malacostraca: Decapoda: Hippolytidae)

Shrimp ranging from Sheep Bay, Alaska, to Baja California, Mexico (Jensen 1995). Size to about 40 mm (Jensen 1995). Color usually green.

Habitat

Found in eelgrass in protected bays, from the low intertidal area to 10 m depth (Jensen 1995). Primarily active at night.

Reproduction

Females of the infraorder Caridea attach eggs to pleopods; eggs do not hatch until they reach a relatively late larval stage (Kozloff 1990). Ovigerous (egg-bearing) females were collected from Elkhorn Slough in April, July, and November (MacGinitie 1935).

***Neotrypaea californiensis* — Bay ghost shrimp**

(Arthropoda: Malacostraca: Decapoda: Callianassidae)

Burrowing shrimp with soft, poorly calcified body and unequal claws. Range from Mutiny Bay, Alaska to Estero Punta Banda, Baja California, Mexico. To 12 cm length. Body with overall orange, pink, or yellowish cast. Although prized as fishing bait and commercially harvested for this purpose, it is considered a pest because its burrowing can bury shell habitat established to encourage reproduction of Dungeness crabs (*Cancer magister*) (Feldman et al. 1997) and can smother or bury oyster beds (Jensen 1995, Dumbauld et al. 1996).

Habitat

Sand and muddy sand of bays and estuaries. Creates branching, impermanent burrows in the intertidal area during the course of deposit-feeding. Burrows range in depth to 50 cm (Swinbanks and Murray 1981) and provide homes for a number of transient and/or specialized commensal species, including species of scale worm (*Hesperonoë complanata*), pea crab (*Scleroplax granulata*), copepod (*Clausidium vancouverense*), and shrimp (*Betaeus* sp.) (Ricketts et al. 1968, Smith and Carlton 1975).

Reproduction

In Willapa Bay, egg-bearing females were collected between April and August; eggs hatched between June and August (Dumbauld et al. 1996). In Monterey Bay, ovigerous females were found year round, but more females carried eggs in June and July than in other months (Morris et al. 1980). Larvae are transported to the nearshore environment by tidal action, where they are planktonic for 6 to 8 weeks (Johnson and Gonor 1982). Long-lived, perhaps reaching an age of 15 to 16 years.

Feeding

Deposit feeder; consumes organic material sorted from the sediment by hairs on the second and third pairs of legs (Morris et al. 1980). Movement of water through the burrow is aided by the beating of paddle-like pleopods.

***Palaemon macrodactylus* — Oriental shrimp**

(Arthropoda: Malacostraca: Decapoda: Palaemonidae)

Exotic shrimp accidentally introduced from southeast Asia in the 1950s during ballast water exchange (Newman 1963). Range from Willapa Bay, Washington to Long Beach Harbor, California; abundant in San Francisco Bay (Jensen 1995). To 55 mm length. Body generally translucent; sometimes green to olive. Often used as fishing bait.

Habitat

Estuaries and brackish waters of tidal creeks. Finds shelter in crevices of complex substrates, in aquatic vegetation, and in depressions in the sediment (Newman 1963). A hardy species that is tolerant to wide variations in temperature and salinity.

Feeding

Opportunistic predator and scavenger. Cannibalism has been observed in laboratory specimens confined to small containers (Newman 1963).

***Hemigrapsus oregonensis* — Yellow shore crab**
(Arthropoda: Malacostraca: Decapoda: Grapsidae)

Small to medium-sized shore crab, ranging from Resurrection Bay, Alaska, to Baja California, Mexico (Jensen 1995). Carapace width to 50mm. Color usually grayish green, although white mottled patterns are common in juveniles; walking legs hairy.

Habitat

Often found under rocks throughout the intertidal zone, especially on gravel and mud beaches, and in estuaries where it constructs burrows in mud banks. Primarily active at night and when burrows are inundated by water. Tolerant of silty conditions and low salinities (Willason 1981). Where they co-occur, interference competition and predation by *Pachygrapsus crassipes* may confine *Hemigrapsus oregonensis* to lower intertidal zones (Willason 1981).

Reproduction

Ovigerous females and newly recruiting juveniles are most abundant from May through August in southern California (Ricketts et al. 1968, Willason 1981).

Feeding

Opportunistic omnivore. Grazes by scraping diatoms from surfaces, cropping algae, scavenging, preying on a wide range of small invertebrates, and even filter-feeding using third maxilliped (Jensen 1995). Preyed upon by striped shore crab, *Pachygrapsus crassipes* (Willason 1981), fishes, such as the longjaw mudsucker *Gillichthys mirabilis* (MacDonald 1975), and wading birds.

***Pachygrapsus crassipes* — Striped shore crab**
(Arthropoda: Malacostraca: Decapoda: Grapsidae)

Small to medium-size crab (to 50 mm carapace width) that ranges from Charleston, Oregon, to Isla de Santa Margarita, Baja California, Mexico (Morris et al. 1980). Characterized by dark coloration (red, purple, or green) and a transversely striated carapace.

Habitat

Abundant in rocky crevices, tidepools, and hard muddy shores of bays and estuaries. *P. crassipes* can tolerate brackish and hypersaline conditions but prefers normal seawater. Individuals spend at least half their time out of water, but they visit pools periodically to feed and moisten their gills (Morris et al. 1980).

Reproduction

Mating occurs just after the female molts (late February to October in southern California) and is preceded by a courtship dance (Hiatt 1948). Peak reproduction is in June and July (Morris et al. 1980). Females produce one or two broods per year, each containing an average of 50,000 eggs (Hiatt 1948). Larvae are planktonic. Juveniles are frequently found on tufts of algae. Adults reach full size 3 years after hatching; females reach sexual maturity at 11 to 12 months after hatching, males at 7 months (Morris et al. 1980).

Feeding

Predominantly herbivorous, but can also be an aggressive, opportunistic carnivore (Hiatt 1948). Typical foods include the algae and diatoms growing on rocks, larger algae (*Enteromorpha*,*Ulva*; *Fucus*), and detritus (Hiatt 1948, Barry and Ehret 1993), but limpets, snails, hermit crabs, and isopods are also occasionally taken (Hiatt 1948, Morris et al. 1980). The excavated tips of the chelae provide an excellent tool for scraping algal mats. *P. crassipes* feeds primarily at night (Hiatt 1948).

Uca crenulata **— Fiddler crab**

(Arthropoda: Malacostraca: Decapoda: Ocypodidae)

Small crab (to 20 mm carapace width) found on sand and mud flats in protected bays and estuaries from Anaheim Bay, California, to Isla de los Mangles, Baja California, Mexico. Females have small chelae of equal size. Males have one small chela and one that is greatly enlarged and used in courtship behavior. The common name for this species, "fiddler crab," stems from the male's motioning of the large chela toward females to attract them into his burrow.

Habitat

U. crenulata form burrows in high and middle intertidal sand and mud flats which are distinguishable by mounds of pellets found around openings (Morris et al. 1980).

Their activity level is directly correlated with soil temperature (i.e., activity is minimal in winter and greatest in summer; Farwell 1966). Burrows range from 10 to 50 cm depth (Kutilek 1968). Females are generally found in wetter areas (at lower elevations), whereas males can be seen in both high and middle intertidal areas (Farwell 1966). *U. crenulata* is tolerant of brackish conditions as well as high salinities, up to 150 ppt (Morris et al. 1980).

Reproduction

Gravid females are found between late July and early September; larvae are released in late summer, remaining pelagic for 1 to 2 months and settling during the fall months (Farwell 1966).

Feeding

Selective deposit-feeder. Feeding generally occurs near burrows (Farwell 1966) and rejected mud can be seen as pellets scattered about the burrows (Morris et al. 1980).

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Appendix 6

Habitat functional requirements for common fish species in southern California salt marshes, lagoons, and estuaries

Gregory D. Williams, Julie S. Desmond, Sharook P. Madon, and Janelle M. West

Acanthogobius flavimanus — **Yellowfin Goby**

Largest gobiid in southern California bays and estuaries (to 290 mm TL; Baker 1979). Native to Japan, Korea, and northern China, it was likely transported to the Pacific coast of the U.S. via ballast water; its range now extends from San Francisco Bay to San Diego Bay (Williams et al. 1998*a*). It has also been collected in Australia (Middleton 1982).

Spawning/reproduction

Age/size at maturity: One year, approximately 101 mm SL (Baker 1979).

Season/timing: January–March (Brittan et al. 1970).

Location/substrate: Estuaries and bays, muddy river bottoms.

Common habitats/movements

Larval habitat/settlement substrate: Larvae are pelagic until 15 to 20 mm TL, then settle on muddy or sandy bottoms.

Adult habitat: Tolerant of a wide range of salinities; yellowfin gobies have been collected from rivers and freshwater canals as well as estuaries (Brittan et al. 1970). Fish seem to prefer muddy substrates (Brittan et al. 1970; Middleton 1982) and in San Diego Bay, they are mainly found in shallow, sometimes vegetated areas (Horn and Allen 1996). Over a 14-year period in San Francisco Bay, yellowfin gobies were collected more often from large sloughs, while native species were more common in smaller, dead-end sloughs (Meng et al. 1994).

Feeding/growth

Behavior: Opportunistic benthic carnivores.

Prey: In the San Francisco Bay-Delta area, yellowfin gobies consumed crustaceans including copepods, mysids, and amphipods; little seasonal variation was observed in the diets (Baker 1979). In San Diego Bay, yellowfin gobies also consumed fish, including juvenile topsmelt and arrow gobies (Desmond, unpub.

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data); piscivory has also been observed in yellowfin gobies collected in Australia (Middleton 1982). Fish of year class 1 and higher appeared to consume a wider variety of prey items than younger fish (Baker 1979).

Fish are thought to reach approximately 101 mm SL by end of year 1, 140 mm SL by end of year 2, and 178 mm by end of year 3 (Baker 1979).

***Atherinops affinis* — Topsmelt**

Pelagic marine fish with preference for bays, estuaries, sloughs, and lagoons. One of the most abundant species in many Pacific Coast estuaries (Allen 1982, Horn and Allen 1985). Geographic range from Gulf of California, Baja California (Mexico) to Tillamook Bay, OR (Emmett et al. 1991)

Spawning/reproduction

Age/size at maturity: Two years, 120 mm TL (Fronk 1969).

Season/timing: Extended period from February–October, with peaks in May–June. Spawning occurs primarily at night, in multiple batches per season (Fronk 1969, Wang 1986)

Location/substrate: Shallow water habitats (tidal flats and small tidal channels) over submerged eelgrass (*Zostera* spp.) and macroalgae (e.g., *Enteromorpha*) (Fronk 1969, Nordby 1982); eggs are benthic and bear filaments which entangle with vegetation (Wang 1986).

Common habitats/movements

Larval habitat/settlement substrate: Larvae (4.3 to 18.5 mm TL) are planktonic but school near the surface in shallow, slow-moving waters over a variety of substrates (Wang 1986).

Juvenile habitat: Juveniles (18.5 to 120 mm TL) are pelagic and found over a wide range of habitats; may venture over intertidal marsh habitats at high tide (West and Zedler *in press*).

Adult habitat: Adults are pelagic and found over a wide variety of habitats. Topsmelt are the dominant species in deeper, high-order, saltmarsh channel habitats (Williams and Zedler 1999).

Movements: A euryhaline species, many adults and juveniles leave to nearshore ocean areas and coastal kelp beds during fall and winter; during late spring, adults move into shallow channels and mudflats to spawn, migrating into upper estuarine areas during summer and fall (Wang 1986).

Feeding/growth

Behavior: Both pelagic and benthic feeding, with peaks during daylight hours, especially during high tides (Hobson et al. 1981, S. Madon, unpublished data).

Prey: Omnivorous; estuarine inhabitats feed on plant material and detritus, while ocean inhabitants are primarily planktonic crustacean carnivores (Fronk 1969, Klingbeil et al. 1975, Hobson et al. 1981). Some dietary shifts are observed through ontogeny: small juveniles (<51 mm TL) feed primarily on epibenthic and planktonic crustacea (copepods, ostracods, amphipods, cumaceans), while larger adults shift to an increasingly detritus and plant-based diet, which often includes the alga *Enteromorpha* spp. (Horn and Allen 1985, G. Williams et al. – unpublished data).

***Clevelandia ios* — Arrow Goby**

A small (max TL = 50 mm) benthic goby commonly found in intertidal to shallow subtidal zones of fresh to marine waters in most Pacific coastal estuaries (Emmett et al. 1991). Geographic range is from the Gulf of California, Baja California (Mexico) to Vancouver Island, British Columbia (Canada) (Miller and Lea 1972).

Spawning/reproduction

Age/size at maturity: Less than one year; 29 mm TL (Brothers 1975).

Season/timing: Spawning occurs all year (depending on estuary), with peaks from February to June (Brothers 1975, MacDonald 1975).

Location/substrate: Intertidal mud or sandflats. The slightly adhesive eggs of arrow gobies are laid on walls of a 10 cm deep burrow (Wang 1986).

Common habitats/movements

Larval habitat/settlement substrate: Larvae are pelagic and widely transported within bays, estuaries, and even offshore (Nordby 1982, Wang 1986). Settlement occurs over a wide range of substrates.

Juvenile and adult habitat: Intertidal and shallow waters of bays, often in burrows of invertebrate commensal hosts. Juveniles prefer bottoms of mixed sand and mud (Prasad 1948).

Movements: Larvae are transported widely. Activity of individuals in the intertidal zone is controlled by the tidal cycle; at low tide, gobies may take refuge within invertebrate burrows.

Feeding/growth

Behavior: Benthic/epibenthic carnivore.

Prey: Primarily small epibenthic crustacea, including harpacticoid and calanoid copepods, ostracods, amphipods, and oligochaetes (Prasad 1948, MacDonald 1975, Barry et al. 1996).

***Fundulus parvipinnis* — California Killifish**

Small (10 to 92 mm TL), common residents of shallow protected bays and salt marshes from Bahia Magdalena, Baja California Sur, Mexico to Morro Bay, California (USA) (Miller and Lea 1972, Swift et al. 1993). Killifish are one of the most common species associated with shallow vegetated saltmarsh and intertidal panne habitats (Allen 1996, Desmond 1996, West and Zedler *in press*).

Spawning/reproduction

Age/size at maturity: Eight months, >46 mm TL (Fritz 1975).

Season/timing: Extended period from April–September, with peaks early in the year (April–June) (Fritz 1975). Repetitive spawning (three peaks) over monthly intervals likely; some evidence of mortality soon thereafter (Fritz 1975).

Location/substrate: Not certain; thought to spawn in intertidal vegetation (e.g., *Spartina*) or permanent pools on marsh surface (Fritz 1975). Spawning timing likely related to tidal cycle. Eggs possess small adhesive strands and clump together; extended exposure to light increases viability of developing embryos (Hubbs 1965).

Common habitats/movements

Larval habitat/settlement substrate: Larvae (6 to 11 mm TL) are considered micronektonic and concentrate in intertidal pools and shallow edge habitats (Fritz 1975, Watson 1996, Williams et al. 1998a).

Juvenile habitat: Juveniles (11 to 46 mm TL) display schooling behavior (Breeder 1959) and are common in shallow, vegetated, edge habitats, especially low-order tidal creeks (Desmond 1996). Juveniles rapidly colonized created habitats at Tijuana Estuary (Williams et al. 1998*b*).

Adult habitat: Shallow habitats in a variety of fresh, brackish, and marine waters of bays and lagoons. California killifish dominated catches in saltmarsh channels with broad, gradually sloping banks, sandy sediments, and shallow waters (Williams and Zedler 1999).

Movements: Most movements appear to be tidally induced; during flooding tides fish move into side channels and follow the rising tide line, while during ebbing

tides fish move from shallows into deeper channels. Seasonal movements are not well known.

Feeding/growth

Behavior: Feeding behavior is highly variable, occurring at the water surface, in the benthos, and throughout the water column. Feeding activity has been correlated to the tides, with peak feeding associated with high tide access to intertidal habitats (Fritz 1975, S. Madon unpublished data). Ontogenetic shifts in the diet of this species have been observed, changing from a juvenile diet dominated by small copepods to larger benthic prey as they mature (Allen 1982, Hartney and Tumyan 1998).

Prey: Most studies indicate that killifish diets are composed predominantly of arthropods (i.e., isopods, tanaids, amphipods, copepods, ostracods, and dipteran insects) but often include a variety of other prey, including annelids, gastropods, and fish ova (Horn and Allen 1985, Fritz 1975, West and Zedler *in press*). During spring tides, killifish are known to forage on vegetated marsh surfaces where they consume six times as much food as in adjacent creek and channel habitats (West and Zedler *in press*). It has been estimated that individuals weigh approximately 4.6 g by the first year (Perez-Espana et al. 1998). Maximum total length is 110 mm (Miller and Lea 1972, Eschmeyer et al. 1983).

***Gambusia affinis* — Mosquitofish**

The mosquitofish is native to the southern Midwest (southern Mississipi River drainage) but has been introduced to virtually every other part of the world as a tool to control mosquito populations (Meffe and Snelson 1989). It is tolerant of poor water quality and high pesticide levels but not able to withstand low temperatures. In California it is found in virtually every low- to mid-elevation fresh and brackish water habitat in the state (Swift et al. 1993). This small fish has had considerable ecological impact on native ecosystems and has been implicated in the elimination or decline of federally endangered and threatened fish and amphibian populations through competition and direct predation on eggs, larvae, and juveniles (Courtenay and Meffe 1989).

Spawning/reproduction

Sexually dimorphic, with males considerably smaller than females. Livebearers (ovoviviparous); males have modified anal fin (gonopodium) to accomplish internal fertilization. Females can store sperm; thus, a single fertilized female can found new populations (Courtenay and Meffe 1989).

Age/size at maturity: Juveniles can mature within a single year.

Season/timing: Multiple batches of young per year, with up to 300 offspring per birth sequence. Reproduction occurs during warmer months.

Common habitats/movements

Mosquitofish are associated with standing to slow moving water in shallow, frequently vegetated, edge habitats; they frequent brackish waters. Juveniles and adults are found in similar habitats. In estuarine habitats of southern California, mosquitofish are found in shallow, brackish waters near tributary inflows. Most commonly collected in marsh habitats during the winter rainy season when water salinities are lowest.

Feeding/growth

Opportunistic omnivores that feed on a wide variety of algal species, invertebrates, including mosquito larvae, and fish eggs and larvae. Aggressive interactions with other fish species have been observed.

Maximum length rarely exceeding 6 cm. Slow growth and reduced reproductive rate at temperatures below 20°C (Vondracek et al. 1988).

***Gillichthys mirabilis* — Longjaw Mudsucker**

A large benthic goby found in most estuaries from Tomales Bay (central California) to Bahía Magdalena (Baja California), and in the Gulf of California (Barlow 1961, Miller and Lea 1972). This species has been reported as far north as Puget Sound (Jordan and Starks 1895).

Spawning/reproduction

Age/size at maturity: ~1 year, 100 mm TL (Barlow 1961).

Season/timing: In the San Diego area, spawning period extends from January–July, and peaks from February–April. (Weisel 1947*a*).

Location/substrate: Eggs are club shaped and attached to a central stalk by adhesive threads, resembling minute yellow grapes on a stem (Weisel 1947*b*). Spawning likely occurs in or near tidal creeks, as Nordby (1982) observed high densities of larvae in this habitat.

Common habitats/movements

Larval habitat/settlement substrate: Larvae are pelagic, typically remaining near the bottom substrate, and seem to prefer areas where tidal velocities are lower (i.e., channels that are distant from river mouths, tidal creeks) (Nordby 1982). The majority of larvae remain in the estuary, while some reach nearshore waters (Barlow 1963, Nordby 1982). Postlarvae settle at about 8 to 12 mm SL. (Barlow 1963).

Juvenile and adult habitat: High intertidal/shallow waters of bays, meso (5 to 18%) to hyper-saline (>40%) (MacDonald 1975). This species typically prefers areas with soft muddy bottoms (Barlow 1961, Barlow 1963) and is usually found in narrow, low-order tidal creek channels characterized by high salinities, low DO levels, and steep, clay banks (Williams and Zedler 1999, Desmond et al. *in press*). They may be commensal in crab burrows along the banks of tidal creeks (MacDonald 1975). Juveniles and adults are also known to use vegetated marsh surface habitats during high spring tides (West and Zedler *in press*).

Movements: Adults are closely restricted to their habitats and generally remain in the marsh-creek system (Barlow 1963).

Feeding/growth

Behavior: Epibenthic carnivore (Barry et al. 1996).

Prey: ghost shrimp, shore crabs, small fish (primarily killifish), isopods, bivalve siphons, amphipods, algae (MacDonald 1975, Macdonald 1977, West and Zedler *in press*).

Within the first year, fish reach a standard length of 100 to 140mm (Barlow 1963).

***Hypsopsetta guttulata* — Diamond Turbot**

A common inhabitant of bays, estuaries, and lagoons. Often the dominant flatfish in southern California bays and estuaries, most common in waters less than 10 m in depth (Lane 1975). Geographic range from Magdalena Bay, Baja California to Cape Mendocino, California, with an isolated population reported in the Gulf of California (Miller and Lea 1972).

Spawning/reproduction

Age/size at maturity: Two to three years, about 180 mm TL (Emmett et al. 1991). Mortality is high immediately after spawning (Lane 1975).

Season/timing: Spawning occurs all year off coastal areas, with a winter peak (November–January) exhibited by populations in southern California, and summer peak (June–October) in northern populations (San Francisco Bay); peak

spawning is likely correlated with temperatures of 14 to 16°C (Eldridge 1975, 1977; Walker et al. 1987).

Location/substrate: Eggs and larvae are primarily planktonic, found over a wide range of substrates (Emmett et al. 1991).

Common habitats/movements

Larval habitat/settlement substrate: Planktonic larvae (1.6 mm SL at hatch) occur in bays, estuaries, and shallow coastal waters within 2 km of shore; larvae metamorphose and settle out of the water column as juveniles on sand and mud bottoms at about 11.0 mm SL (Emmett et al. 1991, Gadomski and Peterson 1988).

Juvenile habitat: Juveniles are found primarily in bays and estuaries on sandy or muddy bottoms (Emmett et al. 1991).

Adult habitat: Adults inhabit bays, estuaries, and nearshore coastal waters down to approximately 152 m but generally prefer depths <4.6 m (Emmett et al. 1991).

Movements: Larvae settle in shallow waters in or near bays and estuaries, and movement within the habitat appears restricted after settling (Lane 1975). Larger fish are known to generally move to lower portions of bays and estuaries, and adults move into coastal areas to spawn (Lane 1975).

Feeding/growth

Larvae are likely planktivores that feed on zooplankton and phytoplankton (Lane 1975, Emmett 1991). Juveniles and adults forage on or in the substrate and appear to feed diurnally, with peak feeding during daylight (Lane 1975). The diet of juveniles and adults includes polychaetes, clams, clam siphons, gastropods, ghost shrimp, amphipods, cumaceans, crustaceans, and small fish (Lane 1975).

Within the first year of growth, fish can reach an average length of approximately 138 mm (77.5 g average weight) (Lane 1975).

***Leptocottus armatus* — Pacific Staghorn Sculpin**

Description and range: Shallow bays and estuaries to offshore waters (intertidal to 300 ft) from San Quintín Bay, Baja to Chignik, Alaska (Miller and Lea 1972). This species can live in riverine, estuarine, and marine environments (Emmet et al. 1991).

Spawning/reproduction

Age/size at maturity: 1 yr, 120 mm TL (Jones 1962).

Season/timing: In southern California, spawning occurs December–March, peaking in January and February (Tasto 1975). Other studies in northern California report an earlier initation of the spawning period (October, Jones 1962; November, Boothe 1967).

Location/substrate: Eggs are demersal and adhesive; substrates include mud, sand, or rock (Wang 1986, Emmett et al. 1991).

Common habitats/movements

Larval habitat/settlement substrate: Larvae (4 to 20 mm TL) are planktonic (Emmet et al. 1991). Settlement occurs at 10 to 15 mm (Jones 1962), ususally in clean sandy habitats in tidal flats and pools (Marliave 1975, Emmett et al. 1991).

Juvenile habitat: Juveniles are typically found in shallow water habitats, such as tidal creeks (Barry et al. 1996), in riverine, estuarine, and marine environments (Emmett et al. 1991).

Adult habitat: Adults generally remain in sandy, highly saline estuarine areas (Wydoski and Whitney 1979).

Movements: Juveniles exhibit seasonal movements within estuaries; in winter months small juveniles settle in estuarine areas, and may move up into freshwater during spring and early summer (Conley 1977). Once spawning is complete, adults likely move into deeper water offshore (Tasto 1975).

Feeding/growth

Behavior: Larvae are planktivorous, adults are primarily epibenthic carnivores (Emmett et al. 1991, Barry et al. 1996). This species feeds continuously throughout the tidal cycle during both day and night (Tasto, 1975, Smith 1980).

Prey: Important predator of ghost shrimp, *Neotrypaea californiensis* (Posey 1986). Dungeness crabs (*Cancer magister*), particularly juveniles (Reilly 1983). All size classes eat various types of shrimp (Armstrong et al. 1995). Adults – fish (usually gobies; Tasto 1975) and large decapod crustaceans. Juveniles – benthic and epibethic organisms, including gammarid amphipods, copepods, polychaetes, bivalve siphon tips (Peterson and Quammen 1982), and juvenile decapod crustaceans (Emmet et al. 1991, Armstrong et al. 1995, Barry et al. 1996).

This species has been reported to live as long as 3 years, reaching a length of 203 cm in California (Jones 1962), while in Washington Wydoski and Whitney (1979) report a lifespan of up to 10 years and a maximum length of 229 mm.

***Mugil cephalus* — Striped Mullet**

Large, schooling fish found in all warm seas, and in the Eastern Pacific from the Galapagos Islands to Monterey. In California it inhabits coastal waters, including estuaries, freshwaters, and the ocean to depths of 400 ft. It makes up a large proportion of the fish biomass in some southern California estuaries (Horn and Allen 1985).

Spawning/reproduction

Age/size at maturity: Two years, 300 mm TL (235 mm SL) (Anderson 1958).

Season/timing: In California, mullet are thought to spawn in the fall and/or winter (Miller and Lea 1972).

Location/substrate: Spawning occurs in surface waters offshore; pre-juveniles and adults move back into the estuaries shortly afterwards (Miller and Lea 1972).

Common habitats/movements

Larval habitat/settlement substrate: Newly spawned larvae (4 to 12 mm TL, Anderson 1958) occur offshore in a wide range of depths, yet are typically found at the surface (Collins and Stender 1989).

Juvenile habitat: Juveniles (116 to 235 mm TL; Anderson 1958) occur in most coastal waters, including estuaries and rivers, over a variety of substrates, particularly sand and mud. In a North Carolina marsh, juvenile striped mullet were most abundant in small, first-order creeks; density decreased in the larger creeks downstream (Weinstein 1979).

Adult habitat: Adults are pelagic and found in a wide variety of habitats, in salinities ranging from fresh to hypersaline. Temperature seems to be more limiting to striped mullet than salinity; in San Diego Bay, the species was absent for several summers in which mean water temperature did not reach 18°C, but reappeared when a warmer series of summers occurred (Radovich 1961).

Movements: Aside from spawning-related movements, striped mullet may also move with tides into expanded foraging areas in the intertidal; they are frequently observed jumping out of the water, although the causes for this are unknown.

Feeding/growth

Striped mullet have a gizzard-like stomach and a long intestine. They feed mainly on algae and detritus, usually taking mouthfuls from the bottom or grazing from submerged plant and rock surfaces (Odum 1970). In the youngest stages (<30 mm), however, they have been found to be carnivorous, feeding on a variety of mosquito larvae, copepods, and zooplankton (Odum 1970). Although feeding has been observed to be nearly continuous, a peak in feeding at high tide was noted by Odum (1970).

Fish reach a standard length of approximately 160 mm at 1 year of age; growth for juveniles has been estimated at 5 mm per month during colder months and up to 17 mm per month during warmer summer months (Anderson 1958).

***Paralichthys californicus* — California Halibut**

An economically important species that is fished commercially and recreationally throughout its range. Overall range from Magdalena Bay, Baja California, to the Quillayute River, Washington (Emmett et al. 1991). Southern California estuaries, bays, and lagoons play a critical role in the early life history of this species (Kramer 1991).

Spawning/reproduction

Age/size at maturity: Males mature at 200 mm TL (2 to 3 years), whereas females mature at 375 mm (4 to 6 years) (Haaker 1975).

Season/timing: Spawning occurs year-round, mostly from January–August; in southern California, spawning occurs from February–July, peaking in May (Emmett 1991).

Location/substrate: Spawning occurs over sandy substrates in shallow coastal areas (Emmett 1991).

Common habitats/movements

Larval habitat/settlement substrate: Larvae are planktonic, about 2 mm at hatching, and are found primarily in shallow water areas in nearshore coastal waters (Ahlstrom and Moser 1975, Ahlstrom et al. 1984). Larvae ≤10 mm long are primarily found between the 12 and 45 m isobaths within 2 to 5 km of shore (Emmett et al. 1991) but are rare in bays and estuaries (Nordby 1982, Kramer 1991). Metamorphosis occurs at a length of 7.5 to 9.4 mm (Emmett et al. 1991), and larvae move inshore as they approach metamorphosis (Kramer 1991). Time to settlement is 5 to 6 weeks at 16°C, and 4 to 5 weeks at 20°C (Gadomski and Caddell 1991).

Juvenile habitat: Small juveniles primarily inhabit bays and estuaries, while larger juveniles (200 to 250 mm) inhabit shallow coastal waters (Haaker 1975, Kramer 1991).

Adult habitat: Adults inhabit nearshore coastal areas, primarily depths of 6 to 40 m, but can be found up to depths of approximately 180 m (Haaker 1975, Plummer et al. 1983, Eschmeyer et al. 1983).

Movements: Larvae occur in a narrow nearshore coastal band; settlement occurs in shallow water areas on the open coast and in bays and estuaries (Haaker 1975, Plummer et al. 1983, Kramer 1991). Longshore currents are likely used to migrate into bays and estuaries, where juveniles reside for 2 to 3 years, after which emigration to the open coast occurs (Kramer 1991). Adults move from deeper waters to about 4 to 6 m depths to spawn (Haaker 1975).

Feeding/growth

Larvae are likely planktivores (Emmett et al. 1991). Ontogenetic shifts in the diet of this species have been observed, with diets of small juveniles consisting of gammarid amphipods, copepods, mysids, shrimp, small fish (primarily gobies), and molluscs, whereas larger juveniles and adults feed primarily on fishes such as northern anchovy, sardines, antherinids, sciaenids, ambiotocids, and occasionally other flatfishes (Haaker 1975, Plummer et al. 1983, Allen, L. G. 1988, Drawbridge 1990). Arrow gobies are a dominant prey for juvenile halibut in estuaries and bays (Haaker 1975, G. Williams and S. Madon, unpublished data).

Juvenile halibut (66 to 130 mm TL) show a diurnal feeding pattern, with peak feeding during the daytime, and consume approximately 26% of their body weight in food ration per day; there is also some preliminary evidence of peak feeding coinciding with low tides, although this area needs further investigation (S. Madon, unpublished data). Laboratory investigations have indicated that small juveniles (42 mm) consume about 59% of their body weight in food ration per day under ad lib feeding conditions (S. Madon, unpublished data).

Halibut are estimated to grow 38 to 88 mm/year and live to 30 years; females exhibit faster growth rates and are generally larger than males (Haaker 1975, Emmett et al. 1991).

***Poecilia latipinna* — Sailfin Molly**

Native to fresh and brackish coastal water drainages of the southern Atlantic Coast of the U.S. and the Gulf of Mexico (Meffe and Snelson 1989). The sailfin molly is tolerant to degraded waters and physicochemical extremes (fresh to hypersaline waters) and has become established in habitats worldwide. It is a common aquarium fish, with many new occurrences attributed to the release of unwanted pets or baitfish. In California, populations have been reported in coastal embayments (Swift et al. 1993, Williams et al. 1998*a*) and in the canals, ditches, and shallow margins of the Salton Sea (Shapovolov et al. 1981, Dill and Cordone 1997).

Spawning/reproduction

Sexually dimorphic; males have colorful, huge, sail-like dorsal fins. Livebearers (ovoviviparous); males have modified anal fin (gonopodium) to accomplish internal fertilization. Males exhibit aggressive courtship behavior and may compete for space or interfere with the reproduction of other species (Williams et al. 1998), including the federally endangered desert pupfishes (Schoenherr 1988).

Females can store sperm; thus, a single fertilized female can found new populations (Courtenay and Meffe 1989).

Season/timing: Multiple batches of young per year, with up to 140 offspring per batch.

Common habitats/movements

Juvenile habitat: In marsh habitats, small juveniles (6 to 13 mm TL) have been observed in shallow puddles on the marsh surface (Williams et al. 1998*a*).

Adult habitat: Adults are common in shallow, first-order marsh creeks, in shallow pools in these creeks, and on the marsh surface. In estuarine habitats, they appear to be much more common where hydrologic regimes (i.e., tidal flushing) have been altered or habitats disturbed (Williams et al. 1998*a*).

Feeding/growth

Adults are primarily herbivorous and eat a variety of algae and detritus.

Mollies attain a maximum total length of 10 cm.

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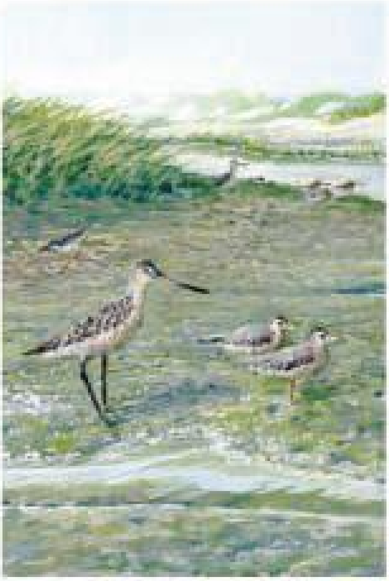

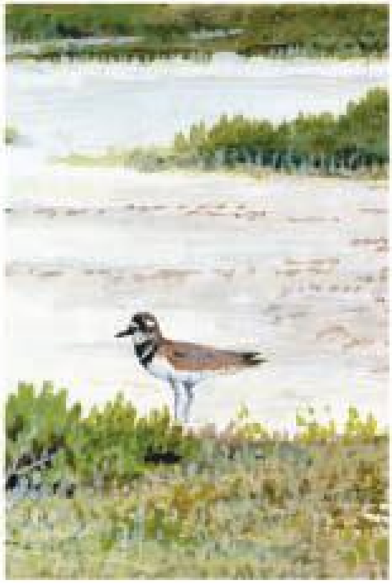

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